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**RESEARCH AND DEMONSTRATION OF IMPROVED METHODS FOR**
**CARRYING OUT BENEFIT-COST ANALYSES OF INDIVIDUAL REGULATIONS**

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**VOLUME III**

**USE OF BENEFIT INFORMATION**
**TO IMPROVE INDIVIDUAL REGULATIONS**

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**PART 6**

**BENEFIT-COST ANALYSIS OF ENVIRONMENTAL REGULATION:**
**CASE STUDIES OF HAZARDOUS AIR POLLUTANTS**

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## I.    INTRODUCTION

The use of benefit-cost analysis in regulating environmental carcinogens and other toxic substances is highly controversial. Economists and other policy analysts typically favor explicit calculation of the costs and the health benefits of regulations, and the use of such estimates in setting standards.  Executive Order 12291, signed by President Reagan in February 1981, encourages this approach by requiring all executive agencies to perform benefit-cost analyses of major regulations and to select, where the relevant statutes permit, those rules that maximize net benefits (42 Fed. Reg. 1319 1981).

Two major lines of criticism have been directed against the use of benefit-cost techniques to evaluate toxic-substance regulations  The first is philosophical, emphasizing such issues as the immorality of making tradeoffs between health and dollars (e.g., Kelman 1981).  We shall not attempt to deal with these philosophical issues in this paper, except to state our belief that such tradeoffs are inevitable, so the relevant question is not whether they should be made, but rather whether they will be made explicitly or implicitly and on what terms.  (For a defense

of the ethical legitimacy of applying benefit-cost analysis to risk-reducing policies, see Leonard and Zeckhauser 1983.)

The second line of criticism stresses practical problems, in particular the difficulty of making quantitative estimates of the benefits of controlling toxic and hazardous substances. Such critics point to fundamental scientific uncertainties about how to estimate the risks from low-level exposures to environmental toxics and to basic disagreements about how much society should be willing to spend to protect health (Swartzman et al. 1982). Many also fear that adding requirements for quantitative assessments of benefits and costs will further delay an already slow regulatory process. Even the debate about these "practical" issues, however, often is remarkably abstract, with little reference to actual decisions that regulators make. Advocates have pointed to the general virtues of quantitative evaluation, while critics have stressed its general unreliability. As a result, these broad debates provide little indication of what is at stake in particular circumstances and, indeed, whether those with very different values and assessments of the scientific evidence might find much common ground in actual regulatory decisions.

This report analyzes the usefulness of benefit assessment and other benefit-cost techniques as they could be applied to decisions the Environmental Protection Agency (EPA) must make in regulating airborne carcinogens under Section 112 of the Clean Air Act (42 U.S.C.A. sec. 7412). The key concepts and findings, however, are of much wider applicability. Although we provide background information on the provisions and history of Section

112 to place our case studies in context, we have not restricted our analysis to regulatory alternatives allowed under the current stature; some of the alternatives that we consider might require statutory changes.

The linchpin of our analysis is the use of benefit information to identify and evaluate alternatives to the technology-based approach EPA has tentatively adopted. We do not analyze cost-based alternatives, such as the "bubble" or marketable permit **schemes.**[1] In particular, we consider the use that can be made of information the EPA has collected on benzene, coke oven emissions, and acrylonitrile -- three pollutants the agency is considering regulating as hazardous substances under Section 112. Information on these three pollutants permits us to provide a rich illustration of the advantages of evaluating benefits explicitly. In addition, the three case studies taken together enable us to assess strategies EPA might employ to establish regulatory priorities and develop regulatory alternatives. Finally, we are able to evaluate the major uncertainties surrounding benefit estimates and to assess how robust our conclusions are when plausible alternative parameter values are used.

We chose Section 112 for our focus both because it is an important (and representative) element in the current framework for regulating environmental toxics and because we were already familiar with some of its key aspects. This report builds upon several earlier studies:  a general evaluation of the advantages of using benefit-cost principles in Section 112 rulemakings

(Harrison 1981); a theoretical study of alternative regulatory strategies that also contains a detailed empirical analysis of benzene (Nichols 1981 and forthcoming); an earlier analysis of coke oven emissions (Haigh 1982); and a study of the merits of varying standards in response to differences in the marginal benefits of controlling emissions (Harrison and Nichols 1983).

The remainder of this chapter summarizes the history of Section 112 and introduces the three case studies. The next chapter presents detailed analyses for the three case studies as well as an overall comparison of regulatory strategies and priorities among the three pollutants. Chapter 3 summarizes the uncertainties in calculating regulatory benefits and analyzes the robustness of the conclusions reached in Chapter 2. Chapter 4 presents our overall conclusions.

## Section 112 of the Clean Air Act

Section 112 was added to the Clean Air Act in 1970 to provide the statutory authority for regulating "hazardous" air pollutants emitted from stationary sources. Hazardous pollutants were to be regulated outside the complex framework of ambient standards, State Implementation Plans, and new source performance standards established for the more ubiquitous "criteria" pollutants. The Act defined a hazardous air pollutant as one

> to which no ambient air quality standard is applicable and which in the judgment of the Administrator causes, or contributes to, air pollution which may reasonably be anticipated to result in mortality or an increase in serious irreversible, or incapacitating reversible, illness. (Section 112(a)(1))

Section 112 requires the Administrator of EPA to establish a list of hazardous air pollutants and, within 180 days of listing a substance, to set emission standards for sources "at the level which . . . provides an ample margin of safety to protect the public health" (Section 112(a)(1)).

This language represented a compromise that emerged from the House-Senate conference committee on the Clean Air Act amendments of 1970.  The Nixon administration had proposed setting national emission standards for hazardous air pollutants based on technological feasibility, while Senator Edmund Muskie and his Democratic colleagues in the Senate favored a zero-discharge requirement (Bonine 1975).  The final language of the section, which refers to neither technological feasibility nor zero discharges, suggests that while the conference committee expected standards to be based solely on health considerations it did not expect health protection to demand absolute elimination of emissions.

   **Dilemmas In Implementation.**  EPA's regulatory activity under Section 112 over the past 13 years has been modest.[2]  Only seven substances have been listed:  beryllium, asbestos, mercury, vinyl chloride, benzene, radionuclides, and inorganic arsenic. Emission standards have been promulgated for just the first four. Both EPA and the environmental groups monitoring the agency's actions under Section 112 have concentrated on pollutants suspected of causing cancer.  This focus on carcinogens has created a dilemma for the agency, as many scientists believe that there are no thresholds for carcinogens -- no exposure levels

short of zero are risk free.  Thus a strict interpretation
of Section 112's requirement to provide "an ample margin of
safety to protect the public health" probably would require zero-
discharge standards, which would be tantamount to banning the
listed substances.  Many of these substances, however, are
important industrial chemicals, so the costs of banning would be
tremendous, with many plant closures and the loss to consumers of
many valuable products.  Instead, EPA has proposed -- and
environmental groups generally have accepted -- standards based
on the degree of control achievable with the "best available
technology" (BAT).  As discussed in more detail below, a
"generic" policy proposed in 1979 would have formalized the
agency's implicit policy of requiring, at a minimum, BAT controls
for sources emitting pollutants listed under Section 112.

     EPA's dilemma and its eventual decision to base control
requirements on technological feasibility are illustrated by the
standards promulgated for asbestos and vinyl chloride.  In 1971,
EPA proposed standards for asbestos because of its link to a form
of cancer known as asbestosis (36 Fed. Reg. 23239 1971).  The
public comments revealed no scientific doubt that asbestos is
hazardous, but they also made clear that a ban would be very
costly because there were few or no alternatives to asbestos in
some of its uses.  Although the EPA maintained that the standard
"was not based on economic considerations" (36 Fed. Reg. 8822
1971) and that "the overriding considerations are health effects"
(36 Fed. Reg. 23239 1971), the preamble acknowledged the dilemma:

        EPA considered the possibility of banning production,
        processing, and use of asbestos or banning all
        emissions...into the atmosphere, but rejected these

approaches...Either approach would result in the
prohibition of many activities which are extremely
important; moreover, the available evidence relating to
the health hazards of asbestos does not suggest that
such prohibition is necessary to protect public health
(36 Fed. Reg. 8820 1971).

The EPA delayed promulgating a final standard until 1973 (well

beyond the 180-day limit) and then only after a court order (38

Fed. Reg. 8820 1973).

The language of the vinyl chloride standard, promulgated in

October 1976 (41 Fed. Reg. 46559 1976), provides an even clearer

indication of the shift to a technology-based approach.  In the

proposed regulation, EPA interpreted Section 112 as allowing it

to set standards

that require emission reduction to the lowest level
achievable by use of the best available control
technology in cases involving apparent non-threshold
pollutants, where complete emission prohibition would
result in widespread industry closure and EPA has
determined that the cost of such closure would be
grossly disproportionate to the benefits of removing
the risk that would remain after imposition of the best
available control technology (40 Fed. Reg. 59534 1975).

Thus, although Section 112 mentions only health effects, and a

literal reading might require that non-threshold pollutants be

banned, the EPA developed an accommodation that based control on

technological feasibility.

Left unresolved in the setting of these individual standards

was the procedure by which substances would be listed.  Asbestos

and vinyl chloride were clear cases of proven carcinogens, but

EPA had identified hundreds of substances as potentially

hazardous air pollutants.  Environmental groups were dissatisfied

with the slow pace at which the agency was listing substances and

promulgating standards.  In 1976 the Natural Resources Defense

Council had entered into a consent decree with EPA in which toxic water pollutants were listed and a schedule for developing regulations was established (Natural Resouces Defense Council, v. Train, No. 75-172, 8 E.R.C. 2120, D.D.C., June 9, 1976). The impetus for considering some overall strategy for airborne toxic substances was provided in November 1977, when the Environmental Defense Fund (EDF) filed a petition requesting that EPA establish the terms of the vinyl chloride agreement as a generic approach to the regulation of all carcinogens (Doniger 1978). In October 1979, EPA proposed a cancer policy entitled "Policies and Procedures for Identifying, Assessing, and Regulating Airborne Substances Posing a Risk of Cancer" (44 Fed. Reg. 58642 1979).

**Cancer Policy.** The proposal was part of a larger effort by the Carter administration to develop regulatory policies for carcinogens. The EPA document was preceded by a controversial cancer policy proposed by the Occupational Safety and Health Administration (OSHA) (45 Fed. Reg. 5001 1980) and by separate policy documents drafted by the President's Office of Science and Technology Policy (1979) and by the U.S. Regulatory Council (44 Fed. Reg. 60039 1979). In addition, the heads of the four major regulatory agencies dealing with carcinogens had formed the Interagency Regulatory Liason Group with a mandate to develop a greater scientific consensus on cancer risk assessment procedures (44 Fed. Reg. 58647 1979). Finally, in 1979 EPA was developing regulations on benzene emissions under Section 112 that could be used as a prototype for the procedure the agency was elaborating in the generic policy. Indeed, when the White House Regulatory

Analysis Review Group (RARG) selected the EPA cancer policy for review, the agency suggested that RARG use benzene as an indication of how the policy would be implemented (Regulatory Analysis Review Group 1980).

The proposed policy never was promulgated. Nonetheless, it is useful to review its provisions because they provide a clear statement of the procedures that had evolved over the first decade of Section 112's existence. The most important features of the document concerned the criteria for listing substances and the criteria for setting standards for source categories (see Harrison 1981 for a more detailed description and critique). The proposal established a relatively low hurdle for listing; any substance that had a high probability of being a carcinogen was to be listed unless it was a "laboratory curiousity." Upon listing, sources emitting the substance immediately would become subject to a set of generic regulations covering maintenance, storage, and various "housekeeping" requirements. For each listed substance, the EPA would prepare detailed estimates of health effects, primarily to set priorities for developing emission standards for individual source categories (e.g., maleic anhdride plants emiting benzene). Those standards were to require, at a minimum, BAT controls. The quantitative risk estimates were not to be employed in the standard-setting process unless they showed that the residual risk after BAT controls would be "unreasonable," in which case tighter controls were to be imposed.

Taken together, these provisions would have created what Harrison (1981) refers to as a "delay trigger" for tight, technology-based regulation of all airborne carcinogens. EPA is aware of the potential for over-regulation that such a policy might create. David Patrick (1982), the chief of the Pollution Assessment Branch in the Office of Air Quality Planning and Standards at EPA, has stated:

> All have perceived that a literal interpretation of Section 112 would not preclude open-ended control requirements or the possibility of zero emission goals, regardless of the control costs. Given this potential and the apparent lack of flexibility regarding the removal of substances from the list of hazardous pollutants or the exclusion of source categories from control requirements, the Agency has also been reluctant to list pollutants as hazardous without some reasonable assurance that subsequent regulations would convey health benefits that are not grossly disproportionate to the costs of control.

In the last two years, the EPA has undertaken considerable analysis of potential Section 112 pollutants, but has not listed any new substances, proposed new standards for substances previously listed, or promulgated standards proposed earlier. The agency does have a list, however, of 37 substances undergoing a variety of studies that might lead to listing and regulation (Patrick 1982).

**Recent Congressional Debate.** The current debate on reauthorization of the Clean Air Act includes Section 112. Environmental groups have criticized EPA for "footdragging," with EDF specifically urging that Congress adopt a generic method for listing airborne carcinogens, list the 37 substances now under study, and require that EPA develop a systematic approach that includes literature reviews, periodic reports, and time limits

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for action.  In contrast, the Chemical Manufacturers Association (CMA) has suggested modifying Section 112 to allow EPA to regulate only those substances that pose a significant risk to health and to consider social, technical, energy, and economic consequences in setting standards (Environment Reporter 1981, 1026).

It is still too early to tell what changes, if any, will be made in Section 112.  Thus far, however, Congressional sentiment, at least in the House, appears to favor swifter, more aggressive regulation of airborne carcinogens.  In August 1982, the House Energy and Commerce Committee voted in favor of an amendment requiring that each year for the next four years, EPA review 25 percent of the 37 substances discussed earlier.  The amendment would establish a presumption in favor of listing; any of the 37 substances automatically would be listed unless EPA determined that it was not hazardous (Environment Reporter 1982, 491). If this provision, or a similar one, is enacted, the pace of regulation under Section 112 should reach substantially higher levels than ever before.

## Introduction to the Case Studies

The pressure from Congress, and environmental groups for accelerating the pace of regulation under Section 112 reflects a concern that potentially serious health threats from airborne carcinogens are going uncontrolled.  To gain a better understanding of the magnitudes of the risks and to examine how benefit-cost techniques, in particular quantitative benefit

assessments, might be used to evaluate and design regulations, we
have selected three substances for detailed study:  benzene, coke
oven emissions, and acrylonitrile.  All three are high-priority
Section 112 pollutants.  Benzene has been listed formally and
regulations have been proposed for several source categories.
Coke oven emissions and acrylonitrile both are on the list of 37
substances, and extensive studies have been performed of their
health risks and control options.

   **Benzene.**[3]  Benzene is a major industrial chemical, ranking
among the top fifteen with a production volume of almost 6
billion kilograms in 1979 (Chemical and Engineering News, June 9,
1980, 36).  Although not counted in production figures, roughly
an equal amount of benzene is found in gasoline (Nichols 1981).
The vast majority of benzene is derived from petroleum, with
relatively small amounts produced as a byproduct of coke ovens.
Most benzene is used to produce other industrial chemicals, which
in turn are used to manufacture a wide range of products
including nylon, plastics, insecticides, and polyurethane foams.
Benzene has long been known as a health hazard at high
concentrations and was regulated under standards covering
hydrocarbons as a class, but until 1977 evidence linking it to
leukemia was considered inconclusive.  In that year, a study by
the National Institute of Occupational Safety and Health (Infante
et al. 1977) showed a much higher than expected incidence of
leukemia in workers exposed to benzene while employed at two
plants in the rubber industry.  The Infante study led OSHA to
reduce its occupational standard and EPA to list benzene under

Section 112.  Other studies provided mixed support for Infante et al.'s results (see Nichols 1981 for a summary of those studies).

Listing led EPA to commission studies of benzene emissions (PEDCo 1977) and exposure (Mara and Lee 1977 and 1978). These studies provided a rough idea of the relative contribution of different types of sources (see Nichols 1981 for a more detailed summary and critique).  Automobiles accounted for over four-fifths of total estimated exposure, although that fraction was expected to fall rapidly as the percentage of automobiles meeting stringent hydrocarbon standards increased.  The next largest source category, service stations, already was subject to vapor recovery controls (for hydrocarbons generally) in most large urban areas.  Moreover, automobiles and service stations involve large numbers of individual sources.  In contrast, the third largest category, chemical manufacturing plants, has few individual sources; the EPA contractor estimated that more than half of all emissions were from the eight plants that used benzene to produce maleic anhydride (PEDCo 1977).  Thus it is not surprising that EPA placed top priority on developing an emission standard for that category.

In April 1980, almost three years after listing benzene, EPA proposed an emission standard for maleic anhydride plants that use benzene as a feedstock (45 Fed. Reg. 26669,1980). The standard would require that existing plants reduce emissions from the main process vent by about 97 percent from uncontrolled levels.  EPA had also considered a 99 percent control requirement, but rejected it on the bases that the "risks remaining after applications of BAT to existing sources are not

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unreasonable" and that the tighter standard might result in the closing of one plant (45 Fed. Reg. 26667 1980). The proposed rule forbids any benzene emissions from new maleic anhydride plants, but higher benzene prices (due to sharp rises in the price of petroleum) already have made it more economical for new plants to use an alternative feedstock (n-butane).

EPA estimated that existing plants operating at full capacity without any controls would emit over 12.5 million kg of benzene annually. A majority of the plants, however, already had controls of 90 percent or better in response to state regulations directed at hydrocarbons or, in the case of one firm, the hope that the benzene recovered would pay for the controls. As a result, full-capacity emissions were estimated at 5.6 million kg per year. The BAT standard proposed was expected to reduce emissions by just under 5.1 million kilograms. Its annual cost (net of credits for the benzene recovered) was put at $2.6 million (U.S. EPA 1980, updated to 1982 dollars).

By early 1981, EPA had proposed benzene standards for three additional source categories: ethylbenzene/styrene plants (45 Fed. Reg. 83448 1980), benzene storage vessels (45 Fed. Reg. 83952 1980), and fugitive sources in petroleum refineries and chemical manufacturing plants (46 Fed. Reg. 1165 1981). All of these categories, however, appeared to be significantly less important sources of benzene emissions than maleic anhydride plants. Our detailed analysis of benzene in the next chapter deals only with maleic anhydride plants.

**Coke Oven Emissions.**[4] Coke, produced by distilling coal in ovens, is essential to the production of iron and steel. In 1979, approximately 48 billion kg of coke were produced in the U.S. (U.S. EPA 1981b). Epidemiological studies of coke-oven workers have shown that emissions from the coking process are carcinogenic, causing increased risk from lung, trachea, bronchus, kidney, and prostate cancers. The toxic elements include both gasses and respirable particulate matter. Most attention has focused on the polycyclic organic matter (POM) contained in the coal tar particulates. Evidence of the carcinogenicity of coke oven emissions from epidemiological studies is supported by animal skin-painting studies, which have found sample extracts to be carcinogenic; by studies showing that laboratory animals exposed to emissions develop lung cancer; by mutagenicity studies with bacteria; and by the fact that numerous constituents of coke oven emissions are known to be carcinogens. In addition to POM, which includes more than 100 individual substances, there is concern about the effects of other substances in coke oven emissons, including aromatic compounds such as benzene; trace metals such as arsenic, berylium, cadmium, lead, and nickel; and gasses such as sulfur dioxide and nitric oxide (U.S. EPA 1982).

Unlike maleic anhydride plants, coke plants have numerous emissions sources. A typical plant contains several batteries, each of which has 20 to 100 ovens. EPA's analyses in preparation for possible listing under Section 112 have focused on three "fugitive" emission sources: doors, topside, and charging leaks.

Charging leaks occur when coal is added to the ovens at the beginning of the coking process. Door leaks are the result of imperfect fits between the ovens and the doors through which the finished coke is later removed.  Topside leaks exit through imperfect seals on the lids and off takes on the tops of the ovens.

Because of the large number of fugitive sources within each plant, EPA believes that monitoring problems would make it impossible to express standards in terms of mass emissions (U.S. EPA 1981b).  Instead, the standards under consideration are stated in terms of visible emissions.  EPA's contractors have identified two alternative levels of control for each source, but cost data are available only for the less stringent of the two, apparently because EPA believes that the tighter standards would result in many plant closures (U.S. EPA 1981).  We expect that if coke oven emissions were listed, standards similar to the following would be specified as BAT:  12 percent leaking doors; 3 percent leaking lids and 6 percent leaking offtakes ("topside"); and 16 seconds of visible emissions for each charging (U.S. EPA 1981a).

Coke oven emissions are regulated at present under OSHA standards (41 Fed. Reg. 46742 1976) and State Implementation plans (SIPs).  Coke-plant operators also have an economic interest in preventing the escape of emissions, as they are used to produce a variety of products (including benzene).  EPA has identified 65 coke plants.  Some plants, however, are closed, others use a dry coal-charging process not covered by the potential regulations, and still others already meet all of the

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standards described above.  As a result, the most recent (April

1983) estimates suggest that only 37 plants would have to

increase control efforts if the standards were imposed (though

some of those plants meet one or two of the three potential BAT

**standards).**[5]  EPA puts the annual control costs for those plants

at $19.3 million (Research Triangle Institute 1983).  Because of

uncertainties about the relationship between visible and mass

emissions, the estimated annual reduction in benzene soluble

organic (BSO) emissions (a reasonable indicator of emissions

posing a threat to health) covers a wide range, from a minimum of

less than 47,000 kg to a maximum of more than 530,000 kg.  For

convenience, most of our calculations employ a simple average of

the minimum and maximum estimates, 289,000 kg of BSO emissions

per year.

**Acrylonitrile.** Acrylonitrile, like benzene, is an important

industrial chemical employed primarily as a feedstock in the

production of other materials that ultimately are used to

manufacture a wide range of common products, including rugs,

clothing, plastic pipes, and automobile hoses.  Almost 1 billion

kilograms of acrylonitrile were produced in 1981 (Chemical and

Engineering News, June 14, 1982).

Evidence of acrylonitrile's carcinogenicity is extensive.

EPA's Carcinogen Assessment Group (CAG) identified three

epidemiological studies; seven lifetime laboratory studies with

rats; several mutagenicity studies with bacteria, Drosophila

(fruit flies), and rodents; chromosomal studies of humans; and numerous metabolic studies.  Respiratory cancers have been associated with acrylonitrile in the epidemiological studies (Albert et al.1982).

EPA has focused on four types of plants that emit significant quantities of acrylonitrile:  (1) acrylonitrile (AN) monomer, (2) acrylic fiber, (3) acrylonitrile-butadiene-styrene (ABS) and styrene-acrylonitrile (SAN), and (4) nitrile elastomer. Plants in the last three categories all use AN monomer as a feedstock.  The largest feedstock use is acrylic fibers, employed primarily to manufacture rugs and clothing. ABS and SAN are both resins used to produce hard plastics for such items as pipes, appliances, disposable utensils, and packaging. Nitrile elastomer is a type of rubber used extensively in the automobile industry for hoses, gaskets, and seals (Radian Corporation 1982).

As with coke ovens, EPA has not yet listed acrylonitrile nor has it proposed specific regulations.  EPA contractors, however, have identified control options for each of the four source categories that we regard as likely candidates for BAT standards. For each source category, these controls would reduce emissions by at least 95 percent below uncontrolled levels.  All of the existing plants, however, already have some controls, so that we estimate that the potential BAT standards would only cut acrylonitrile emissions from 3.6 million kg to 0.5 million kg, a reduction of slightly less than 87 percent.  Estimated annual control costs for the four industries total almost $29 million (updated to 1982 dollars).[6]

**Summary.** Each of the proposed or potential BAT regulations would prevent the release of at least several hundred thousand kilograms of carcinogenic material into the air each year. Moreover, we believe that it would be extremely difficult to argue that any of the controls contemplated are beyond the technological or financial capabilities of the industries affected; in each case some existing plants already meet the standard and in no case do the estimated control costs exceed about 2 percent of total production costs.[7] Indeed, a plausible case could be made that the language of Section 112 calls for tighter standards than those being considered. With the data presented in this chapter, however, it remains an open question whether the health benefits provided are likely to be commensurate with the costs.

## II.  BENEFIT-COST ANALYSES OF THE CASE STUDIES

None of the standards in the three case studies is costly enough to qualify as a "major" rule under E.O. 12291, and thus none would require preparation of a complete benefit-cost analysis.[8]  The data assembled by EPA, however, are sufficient to allow us to make some crude estimates of both the benefits and the costs.  In addition to using this information to evaluate the uniform BAT standards, we also consider two alternative approaches:  (1) modification of the uniform standards to increase net benefits and (2) differential standards based on exposure levels around individual plants.

We begin with a brief, general overview of the steps involved in estimating the benefits of regulating airborne carcinogens.  To illustrate the details of our calculations, we provide a full analysis of the case study of benzene emissions from maleic anhydride plants.  We then summarize the results for coke ovens and acrylonitrile.  The final section compares the three case studies along several dimensions.

### Steps In Estimating Benefits

To estimate the benefits of the standards, we need to trace through the links from emissions to exposure to risk, and ultimately we must estimate the dollar value of reducing risk. Figure 1 presents the steps in schematic form.  These steps are common to all three of the case studies, and indeed to estimating the benefits of controlling virtually any health-threatening

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┌──────────────┐      ┌──────────────┐      ┌──────────────┐      ┌──────────────┐
│              │      │              │      │              │      │              │
│  Emissions   │─────▶│   Exposure   │─────▶│     Risk     │─────▶│    Dollar    │
│              │      │              │      │              │      │  Valuation   │
│              │      │              │      │              │      │              │
└──────────────┘      └──────────────┘      └──────────────┘      └──────────────┘
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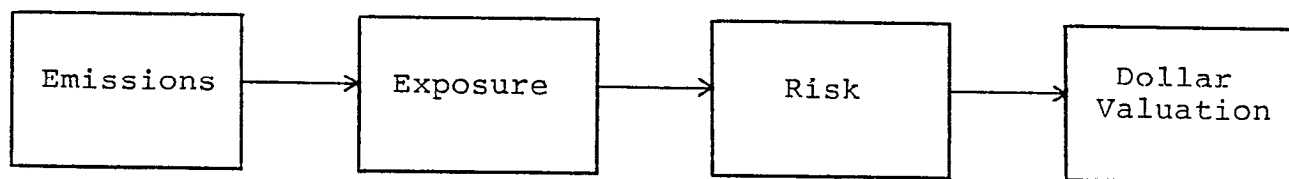
Figure 1.   Steps  on  Estimating  Benefits

pollutant.  We deal briefly with each in turn.  We shall return in the next chapter to discuss the uncertainties associated with each step.

**Exposure.**  Estimating the effect of reduced emissions on exposure requires estimating both dispersion and population patterns.  In each case, EPA has performed general dispersion modeling for a "model plant."  For a given level of emissions, the dispersion model generates estimates of average annual concentrations at various distances from the source.  The modeling for coke ovens and acrylonitrile was carried out to 30 kilometers (km), but the maximum distance used for maleic anhydride plants was 20 km.  (In the next chapter, we show that the shorter distance for maleic anhydride plants is unlikely to make much difference in the results.)  In all three cases, EPA's dispersion modeling did not account for intersource differences in meteorological conditions.

The estimated concentrations then may be combined with plant-specific population data for each source to estimate total exposure levels for a given level of emissions.  For consistency we summarize these exposure levels in terms of **"ug/m$^3$-person-years**," which is simply the average annual concentration (in micrograms per cubic meter) multiplied by the number of people exposed and the length of exposure.  Thus, for example, 1000 people exposed, on average, to 10 **ug/m$^3$** for one year generates 10,000 **ug/m$^3$-person-years** of exposure, as does 10,000 people exposed to 1 **ug/m$^3$.**  As we discuss below, with a linear dose-response model, in which risk is proportional to exposure, this

summary statistic provides sufficient information to predict
total risk; the total risk is independant of how total exposure
is distributed.

Dividing total exposure by the emissions level gives what
Nichols (1981) calls an "exposure factor," the amount, of exposure
caused by a unit of emissions from a particular source.  These
exposure factors, which are measured here in $ug/m^3$-person-years
per kilogram emitted, make it easy to convert estimates of
emissions reductions into more meaningful estimates of exposure
reductions for individual plants.  If a plant with an exposure
factor of 0.6 $ug/m^3$-person-years/kg reduces its emissions by 1
million kilograms, for example, exposure falls by 0.6(1,000,000)
= 600,000 $ug/m^3$-person-years.

**Risk.**  Translating reduced exposure into reduced risk
requires estimating the unit risk factor for each substance.
Typically evidence of carcinogenicity comes from either high-dose
animal studies or from epidemiological studies of workers exposed
to relatively high concentrations of the substance.  The doses at
which risk has been measured often are 1000 or more times higher
than the exposure levels affected by the regulations.  A variety
of mathematical models has been proposed for extrapolating from
high to low doses; scientists disagree as to their validity and
unfortunately they yield wildly different estimates of low-dose
risks.  There is also considerable controversy about how animal
data should be used to predict human risks. Fortunately, we have
human epidemiological data for all three substances.

In each case, we have relied on unit-risk estimates prepared by EPA's Carcinogen Assessment Group (CAG). The CAG employs a procedure that in essence assumes that risk is proportional to dose at low levels of exposure. This model is the most conservative of the models generally used; most scientists accept it as providing an upperbound estimate of the risks from low levels of exposure.

Table 1 presents the unit risk estimates for the three substances. For consistency, we present each as the risk of cancer per $ug/m^3$-person-year. The benzene risk estimate of 1.1 x $10^{-7}$, for example, implies that if exposure to benzene were reduced by 10 million $ug/m^3$-person-years, roughly 1 case of leukemia would be averted. In contrast, reducing exposure to coke oven emissions by the same amount would prevent more than 100 cases if cancer. Thus, although all three substances are carcinogens, their relative potencies vary by more than a factor of 100; coke oven emissions appear to be far more dangerous than either benzene or acrylonitrile.

**Valuing Reduced Risk.** Benefit-cost analysis requires that the costs and benefits be expressed in the same units, usually dollars. For our three case studies, that means that we must assign a value to "saving a life," as the primary risks being avoided are those of cancers with high motality rates. Over the past decade or two, a substantial literature has grown up around the issue of valuing reductions in  risks to life. A consensus appears to have emerged, at least among economists, that the appropriate criterion is the standard one of willingness to pay.

Table 1.  CAG Unit Risk Factors

| Substance | Reference | Unit Risk $(cancers/ug/m^3-yr)$ |
|---|---|---|
| Benzene | Albert et al (1979)[a] | $1.1 \times 10^{-7}$ |
| Coke oven emissions | U.S. EPA (1982)[b] | $1.3 \times 10^{-5}$ |
| Acrylonitrile | Albert et al. (1982)[c] | $4.4 \times 10^{-7}$ |

Notes:

[a]Converted from ppb-person-years to $ug/m^3$-person years.

[b]Converted from lifetime to annual risk.

[c]Average of three risk estimates, converted from lifetime to annual risk.

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The principle is a simple one:  the value of some benefit to an individual is the amount he would be willing to pay to secure it.[9]  (A slightly different formulation, which should yield virtually identical results when dealing with small risks, is to ask how much money an individual would have to receive to forgo the benefit.[10])

Several techniques have been suggested for estimating willingness to pay, including direct questions and drawing inferences from actual behavior.  Economists generally have felt more confident about the latter approach, and more than five studies have estimated the wage premiums associated with occupational risks.  One study (Blomquist 1977) estimated the value of risk reduction using data on automobile seatbelt use.

Bailey (1980) has reviewed the major empirical studies, adjusting them for consistency.  He estimates a range of $170,000 to $175,000 per life saved, with an intermediate estimate of $360,000 in 1978 dollars.  (Adjusting for inflation, as measured by the implicit GNP price deflator, his intermediate estimate translates to $500,000 in 1982 dollars.)  His estimates are based on the work of Thaler and Rosen (1974), Blomquist (1977), and Dillingham (1979).  Two other studies, by Smith (1976) and Viscusi (1978), estimated much higher wage premiums for occupational risks, with the highest estimates in excess of $5 million per life saved in 1982 dollars, but little support can found in the literature for using values as high as that for evaluating programs to reduce risk.

**Value of Exposure Reduction.**  Combining the CAG unit risk factors and the values per life saved discussed above, we can estimate the marginal benefits of reducing exposure to the three substances.  Table 2 presents the results for values per life saved ranging from $250,000 to $5 million.  Each entry is simply the CAG risk factor times the value per life saved. The entry for benzene and a value per life saved of $1 million, for example, is **(1.1 x 10$^{-7}$ deaths/ug/m$^3$**-person-year) ($1 x 10$^6$/death) **= $0.11/ug/m$^3$**·-person-year.  These estimates should be used with caution,as they rely solely on the CAG risk estimates, which may be biased upwards.  In each case, the plausible range includes zero as a lower bound, as many of the nonlinear dose-response models would predict infinitesimal risks for these substances at low concentrations.

## Maleic Anhydride (Benzene) Case Study[11]

As discussed earlier, the only one of our cases for which a standard actually has been proposed is benzene emissions from maleic anhydride plants.  The standard proposed for existing plants would limit them to 0.3 kg of benzene emitted per 100 kg of benzene input, roughly a 97 percent reduction from uncontrolled levels.  Of the eight plants identified by EPA as using benzene, however, five already had controls of 90 percent or better prior to the proposal of the standard.  Indeed, three already met or exceeded the 97 percent requirement.  Thus, EPA's figures suggested that the standard would affect only five plants, and that for two of them the incremental reduction in

Table 2.  Value of Exposure Reduction

| Substance | Value Per Life Saved ($1 million) | | | | |
|---|---|---|---|---|---|
| | 0.25 | 0.5 | 1.0 | 3.0 | 5.0 |
| Benzene | 0.028 | 0.055 | 0.11 | 0.33 | 0.55 |
| Coke oven emissions | 3.25 | 6.50 | 13. | 39. | 65. |
| Acrylonitrile | 0.11 | 0.22 | 0.44 | 1.32 | 2.20 |

Note:
    Entries are value of exposure reduction, in $/ug/m$^3$-person-year.

emissions would be relatively minor due to preexisting controls. In most cases plants had installed controls to meet state regulations, though in one or two cases firms had hoped that the controls would pay for themselves in terms of the benzene recovered for reuse.

Table 3 lists the eight plants, their existing controls, and EPA's estimates of the costs of meeting the two alternative control levels considered, 97 and 99 percent. No costs are shown for plants that already met the proposed standard. For the two plants that had 90 percent controls, however, the cost estimates assume that they would need all-new control equipment; no credit is given for possible adaptation of existing controls. All of the cost estimates are for carbon adsorption, which the EPA estimates indicated would be the lowest-cost control technique (including a credit for benzene recovered), and all assume 100 percent capacity utilization. (The EPA estimates indicate that at lower levels of operation, costs actually rise slightly.)

The costs shown in Table 3 are quite modest, measured relative either to total sales of maleic anhydride or to the costs of most EPA control regulations. It would be extremely difficult to argue that the standard would be "unaffordable" or that it goes beyond widely applied existing technology. We would argue, however, that the cost estimates are meaningless in isolation, that they can judged appropriately only in relation to the benefits they secure.

Table 3.  Control Costs for Maleic Anhydride Plants

| Plant | Current control (%) | Control Costs[a] ($1000/year) | |
|---|---|---|---|
| | | 97% | 99% |
| Ashland | 0 | 520 | 539 |
| Denka | 97 | 0[b] | 468 |
| Koppers | 99 | 0[b] | 0[c] |
| Monsanto | 0 | 687 | 710 |
| Reichold(IL) | 90 | 406 | 422 |
| Reichold(NJ) | 97 | 0[b] | 311 |
| Tenneco | 0 | 270 | 284 |
| U.S. Steel | 90 | 694 | 716 |
| Total | | 2577 | 3451 |

Source:  U.S. EPA, 1980, tables 5-5a to 5-6a, updated to 1982
dollars using GNP implicit price deflator.

Notes:  [a]All cost estimates assume "full-capacity" operation.

[b]Plant already meets 97 percent standard.

[c]Plant already meets 99 percent standard.

The first two columns of numbers in Table 4 report the estimated plant-specific reductions in emissions at the two alternative control levels. The center column shows the estimated exposure factor for each plant. The final two columns present the estimated reductions in exposure; they are simply the emissions reductions multiplied by the exposure factors. Two facts stand out immediately: (1) the plants vary widely in their exposure factors, from a low of only 0,026 $ug/m^3$-person-years/kg to a high of 1.23 $ug/m^3$-person-years/kg, a range of almost a factor of 50; (2) one plant, Monsanto, accounts for over 80 percent of the reduction in exposure at either control level, though it is responsible for less than half the reduction in emissions and only about one-quarter the control costs. Combining the reduction in exposure with the CAG unit-risk estimate yields the prediction that the proposed standard would eliminate $(3.65 \times 10^6 \ ug/m^3$-person-years$)(1.1 \times 10^{-7}$ deaths/$ug/m^3$-person-year $= 0.4$ deaths per year.

**Cost-Effectiveness of Proposed Standard.** Table 5 summarizes the effects of the two standards considered by EPA. The entries under "Annual Costs and Benefits" are simply the totals from Tables 3 and 4. The "Cost-Effectiveness" ratios are the costs divided by the relevant benefits. EPA's proposed standard, 97 percent, has a ratio of $0.71 per $ug/m^3$-person-year of exposure reduction; that is, for the proposed standard to yield positive net benefits, the benefit per unit of exposure reduction, V, would have to exceed $0.71, which is well above the plausible range shown in Table 2. (Even with the CAG risk

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Table 4.   Reductions in Emissions and Exposure for
Maleic Anhydride Plants

| Plant | Emissions Reduction (1000 kg/year) 97% | 99% | Exposure Factor (ug/m$^3$-years) /kg) | Exposure Reduction (1000 ug/m$^3$-years) 97% | 99% |
|---|---|---|---|---|---|
| Ashland | 1722 | 1788 | 0.054 | 93.0 | 96.6 |
| Denka | 0 | 18 | 0.794 | 0. | 14.3 |
| Koppers | 0 | 0 | 0.518 | 0. | 0. |
| Monsanto | 2412 | 2505 | 1.251 | 3018. | 3134. |
| Reichold(IL) | 61 | 110 | 0.026 | 1.6 | 2.9 |
| Reichold(NJ) | 0 | 11 | 1.229 | 0. | 13.5 |
| Tenneco | 747 | 776 | 0.624 | 466. | 484. |
| U.S. Steel | 117 | 211 | 0.573 | 67. | 121. |
| Total | 5059 | 5418 | | 3646. | 3866. |

Source:   Calculated from U.S. EPA (1980).

Table 5.  Cost-Effectiveness of Uniform Emission Standards
for Maleic Anhydride Plants

| | Change in Control Level | | |
| | Current to 97% | Current to 99% | 97% to 99% |
| --- | --- | --- | --- |
| **Annual Costs and Benefits** | | | |
| Control Cost ($1000) | 2577 | 3451 | 874 |
| Reduced Emissions (1000 kg) | 5059 | 5418 | 359 |
| Reduced Exposure (1000 $ug/m^3$-yrs) | 3646 | 3866 | 220 |
| **Cost-Effectiveness** | | | |
| Emissions ($/kg) | 0.51 | 0.64 | 2.43 |
| Exposure ($/ug/$m^3$-yr) | 0.71 | 0.89 | 3.97 |
| Lives saved ($1 million/life) | 6.5 | 8.1 | 36.1 |

estimate, one would have to value "saving a life" at $6.5 million to justify the standard.)  The average ratio for the 99 percent standard is only slightly higher, $0.89 per ug/m$^3$-person-year, but the marginal ratio, which is the relevant one for decision-making purposes, is much higher:  dividing the incremental costs of the tighter standard by the incremental reduction in exposure yields a cost of $3.97 per ug/m$^3$-person-year, almost ten times higher than our upperbound estimate of the marginal benefit.

   **Improving the Uniform Standard.**  The high cost per unit of benefit under the proposed standard is due in part to the fact that of the five plants that would need new control equipment to meet it, two already achieve 90 percent control.  Relaxing the standard to that level would allow those plants to use their existing controls and would save a considerable amount of money with little change in benefits.  Unfortunately, EPA has not developed cost estimates for 90 percent controls.  We can make a conservative estimate of the net benefits of relaxing the standard by assuming that 90 percent controls would cost just as much as those achieving 97 percent for the three plants that currently are uncontrolled.  That is, we assume that with a less stringent standard, Ashland, Monsanto, and Tenneco would cut emissions by only 90 percent, but their costs would be the same as at 97 percent.  Table 6 reports the results (assuming full-capacity operation).  The estimated exposure reduction is only 6.4 percent lower than at 97 percent, but costs fall 42.7 percent.  The cost-effectiveness ratio drops to $0.43 per ug/m$^3$-person-year of exposure reduction, which, while probably still

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Table 6.  Cost-Effectiveness of Less Stringent Standards
for Maleic Anhydride Plants

| | Chance in Control Level | |
| | Current to 90% | 90% to 97% |
| --- | --- | --- |
| **Annual Costs and Benefits** | | |
| Control Cost ($1000) | 1477 | 1099 |
| Reduced Emissions (1000 kg) | 4646 | 413 |
| Reduced Exposure (1000 $ug/m^3$-yrs) | 3405 | 240 |
| **Cost-Effectiveness** | | |
| Emissions ($/kg) | 0.32 | 2.66 |
| Exposure ($/$ug/m^3$-yr) | 0.43 | 4.58 |
| Lives saved ($1 million/life) | 3.9 | 41.6 |

too high to justify the 90 percent standard on benefit-cost grounds, is a substantial improvement over the BAT proposal.  The 90 percent standard yields higher net benefits for all values of V less than $4.58 (roughly $42 million per life saved).

**Differential Standards.**  Relaxing the uniform standard to 90 percent improves cost-effectiveness by screening out plants where the proposed standard has little impact on emissions or exposure. Differential standards, setting tighter requirements for plants with high exposure factors, offers a more ambitious and controversial way of increasing efficiency.  (See Harrison and Nichols 1983 for a general discussion of the advantages of varying standards in response to inter-plant differences in the marginal benefits of emission control.)  As discussed earlier, one plant (Monsanto) accounts for a large fraction of the benefits of the proposed standard.  In part that reflects the fact that it is the largest plant without current controls  More important, however, is that Monsanto has the highest exposure factor; located in a major city (St. Louis), each kilogram emitted from the Monsanto plant causes substantially more exposure, and thus risk to health, than a kilogram emitted by most other plants.  (The only other plant with a similarly high exposure factor, the Reichold plant in New Jersey, already achieves 97 percent control.)

In extreme form, differential standards based on exposure factors lead to plant-specific standards, as each plant has a different exposure factor.  A more practical approach is to establish a limited number of categories; in this case, with only

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eight plants, it is difficult to justify more than two classes.
If we split them evenly, based on the exposure factors in Table
4, there are four "high-exposure" plants [Monsanto, Reichold
(NJ), Denka and Tenneco] and four "low-exposure" plants
[Koppers, U.S. Steel, Ashland, and Reichold (IL)]. Table 7
presents the results for each group.  In the high-exposure group,
97 percent is justified if V exceeds $0.27 (90 percent has a
worse cost-effectiveness ratio), and tightening the standard to
99 percent yields positive net benefits only if the marginal
benefit of controlling exposure exceeds $5.  In the low-exposure
group, even 90 percent control requires a V of more than $5.78.
(The next step for low-exposure plants is 99 percent, which has a
lower cost-effectiveness ratio than 97 percent control.)
Imposing 97 percent control only on the high-exposure plants,
with no new controls on the other plants, yields 96 percent of
the benefits of the proposed uniform standard at 37 percent of
its cost.  That conditional standard dominates the uniform 90
percent alternative, achieving slightly greater benefits at 71
percent of its cost.  Thus, even a crude, two-level approach
significantly improves the cost-effectiveness of standards here.

　　We also can consider the effects of varying the dividing
line between high- and low-exposure plants.  Table 8 ranks the
five plants that do not already meet the 97 percent standard in
descending order of their exposure factors.  It also reports the
cost and benefits of controlling each plant and all other plants
with higher exposure factors.  The final two columns show the
average and marginal costs of exposure reduction.  The latter

Table 7.  Cost-Effectiveness of Conditional Standard for
Maleic Anhydride Plants

|  | High Exposure | | Low Exposure | |
|---|---|---|---|---|
|  | 97% | 99% | 90% | 99% |
| **Annual Costs and Benefits** | | | | |
| Control Cost ($1000) | 957 | 1773 | 520 | 1678 |
| Reduced Emissions (1000 kg) | 3007 | 3159 | 1639 | 2109 |
| Reduced Exposure (1000 $ug/m^3$-yrs) | 3485 | 3645 | 90 | 221 |
| **Marginal Cost-Effectiveness** | | | | |
| Emissions ($/kg) | 0.32 | 5.37 | 0.32 | 2.46 |
| Exposure ($/$ug/m^3$-yr) | 0.27 | 5.10 | 5.78 | 8.84 |
| Lives saved ($1 million/life) | 2.5 | 4.4 | 52.5 | 80.4 |

Table 8.  Effects of Varying Definition of High-Exposure Class
          for Maleic Anhydride Plants

| Marginal Plant | Exposure Factor | Cumulative Cost ($1000) | Cumulative Exposure Reduction | Cost-Effectiveness ($/ug/m$^3$-year) | |
|---|---|---|---|---|---|
| | | | | Average | Marginal |
| Monsanto | 1.251 | 687 | 3018 | 0.23 | 0.23 |
| Tenneco | 0.624 | 957 | 3484 | 0.27 | 0.58 |
| U.S. Steel | 0.573 | 1651 | 3551 | 0.46 | 10.36 |
| Ashland | 0.054 | 2171 | 3644 | 0.60 | 5.59 |
| Reichold (IL) | 0.026 | 2577 | 3646 | 0.71 | 203. |

range from a low of **$0.23/ug/m$^3$-person-year** if only Monsanto is included in the high-exposure class to a high of over $200 if all plants are regulated at 97 percent (at which point, of course, we again have the uniform standard proposed by EPA).

**Recent Developments.** The analysis above is based on data available to EPA when it proposed the standard for maleic anhydride plants in April 1980. Since that time, while the standard has been awaiting promulgation, several important developments have occurred: (1) four plants -- Koppers, Tenneco, and both Reichold facilities -- have shut down; (2) Ashland and Denka have converted to n-butane, apparently in response to higher benzene prices; (3) Monsanto has installed 97 percent controls and is in the process of converting all of its capacity to n-butane; and (4) an additional plant, a small one operated by Pfizer in a lightly-populated area of Indiana, has been "discovered" (Nichols forthcoming). As a result, the costs and benefits of the proposed standard both are smaller, but the latter have fallen by a much larger factor. The standard, if promulgated, would apply to only two plants, Pfizer and U.S. Steel. The former, by virtue of its small size and low exposure factor (0.052 **ug/m$^3$-person-years/kg)** has an estimated cost per life saved in excess of $60 million (Nichols 1981, converted to 1982 dollars). As reported earlier, U.S. Steel has an even worse cost-effectiveness ratio because it already achieves 90 percent control. Thus it appears impossible to justify additional controls for any plants.

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## Coke Ovens Case Study[12]

The standards being considered for coke ovens, as discussed in Chapter 1, are expressed in terms of visible emissions from three sources within coke plants: doors, topside, and charging. Many plants already meet some or all of the requirements: approximately 37 would have to undertake additional action to control at least one of the three sources. All of EPA's emission and cost estimates are from a baseline that assumes compliance with existing state and OSHA regulations. As discussed in Chapter 1, EPA has prepared minimum and maximum estimates of emissions for each plant; here we use the average of those extremes. (In Chapter 3, we consider the effects of using alternative emission-reduction estimates.)

Estimated exposure factors for the coke plants cover an even wider range than those for maleic anhydride plants, from a low of 0.058 to a high of 5.93 $ug/m^3$-**person-years** per kilogram of emissions (calculated from U.S. EPA 1981b, app. E). The mean (weighted by emissions reductions) is 2.8 $ug/m^3$-**person-years/kg.** These exposure factors generally are higher than those for maleic anhydride plants, despite the fact that the same basic dispersion model and meteorological data were used. At least two factors contribute to the differences: (1) coke plants are located in areas with higher population densities and (2) fugitive emissions, released at lower heights with lower exit velocity, apparently result in higher concentrations in the areas around the plants.

**Cost-Effectiveness of BAT Standard.** The first column of
Table 9 summarizes the effects of the BAT standard being
considered by EPA. As reported earlier, the estimated costs are
$19.3 million per year, with emissions reduced by just under
290,000 kg annually. Using plant-specific emission estimates and
the exposure factors described above, we estimate that exposure
would fall by 819,000 $ug/m^3$-person-years. Thus the cost-
effectiveness ratio is about $23.6 per $ug/m^3$-person-year,
substantially higher than that for maleic anhydride plants. It
is important to remember, however, that coke oven emissions are
much more potent carcinogens than benzene; the estimated cost per
life saved is $1.8 million, less than one-third the ratio
estimated for maleic anhydride plants. Nonetheless, we suspect
that most benefit-cost analysts would conclude that the BAT
standard would not yield positive net benefits.

**Improving the Uniform Standard.** With the data available to
EPA, it is possible to consider the individual components of the
BAT standard, though not to analyze alternative levels for the
different sources within plants. The last three columns of Table
9 present separate figures for doors, topside, and charging.
Controls on charging appear to be substantially less cost-
effective than those for the other two sources: the cost per unit
of exposure reduction is over $71, implying a cost per life saved
of $5.5 million. Dropping the charging standard reduces costs by
29 percent, but cuts benefits by only 9 percent. The most cost-
effective component is that for doors, with a cost-effectiveness

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Table 9.   Cost-Effectiveness of Uniform Emission Standards
          for Coke Ovens

|  | Total | Individual Control Options | | |
|---|---|---|---|---|
|  |  | Doors | Topside | Charging |
| **Annual Costs and Benefits** | | | | |
| Control Cost ($1000) | 19,303 | 11,730 | 2,068 | 5,505 |
| Reduced Emissions (1000 kg) | 289 | 230 | 32 | 26 |
| Reduced Exposure (1000 $ug/m^3$-yrs) | 819 | 660 | 88 | 71 |
| **Cost - Effectiveness** | | | | |
| Emissions ($/kg) | 67 | 51 | 64 | 209 |
| Exposure ($/ug/m^3$-yr) | 23.6 | 17.8 | 23.4 | 71.3 |
| Lives saved ($1 million/life) | 1.8 | 1.4 | 1.8 | 5.5 |

Sources:

    Cost data:  Research Triangle Institute, 1983.
    Emission data:  average of "maximum" and "minimum" estimates
       in U.S. EPA 1983b.
    Exposure factors:  calculated from U.S. EPA 1981b, app. E.

ratio of less than \$18 per $ug/m^3$-person-year.  Imposing the

standard just on doors would cut costs 39 percent while still

yielding 80 percent of the benefits of the complete BAT standard.

Note, however, that even the door standard fails to yield

positive net benefits unless the value ascribed to saving a life

is at least \$1.4 million (based, again, on the CAG risk

estimate).

**Differential Standards.**  The wide range in exposure factors

offers opportunities to increase efficiency by restricting the

standards, or portions of it, to plants with relatively high

exposure factors.  Of the 37 plants, for example, 21 have

exposure factors greater than 2.0 $ug/m^3$-person-years/kg.

Imposing the door and topside standards only on those plants

yields 81 percent of the benefits (666,000 $ug/m^3$-person-years) at

only 33 percent of the cost (\$6.3 million) of the uniform BAT

standard.

Figure 2 plots the full range of possibilities for

exploiting differences among plants in exposure factors.  The

axes measure cumulative costs and exposure benefits as

percentages of the maximum levels achieved under the full set of

uniform BAT standards.  For comparison, we show the three

segments of the alternative uniform standards, representing

controls of the three separate sources.  The "differentials

standards" option was derived by first ranking the plants by

exposure factors.  We then computed the cost per unit of exposure

reduction for each of the three control options for each plant.

Finally we found the convex set of plant-control combinations
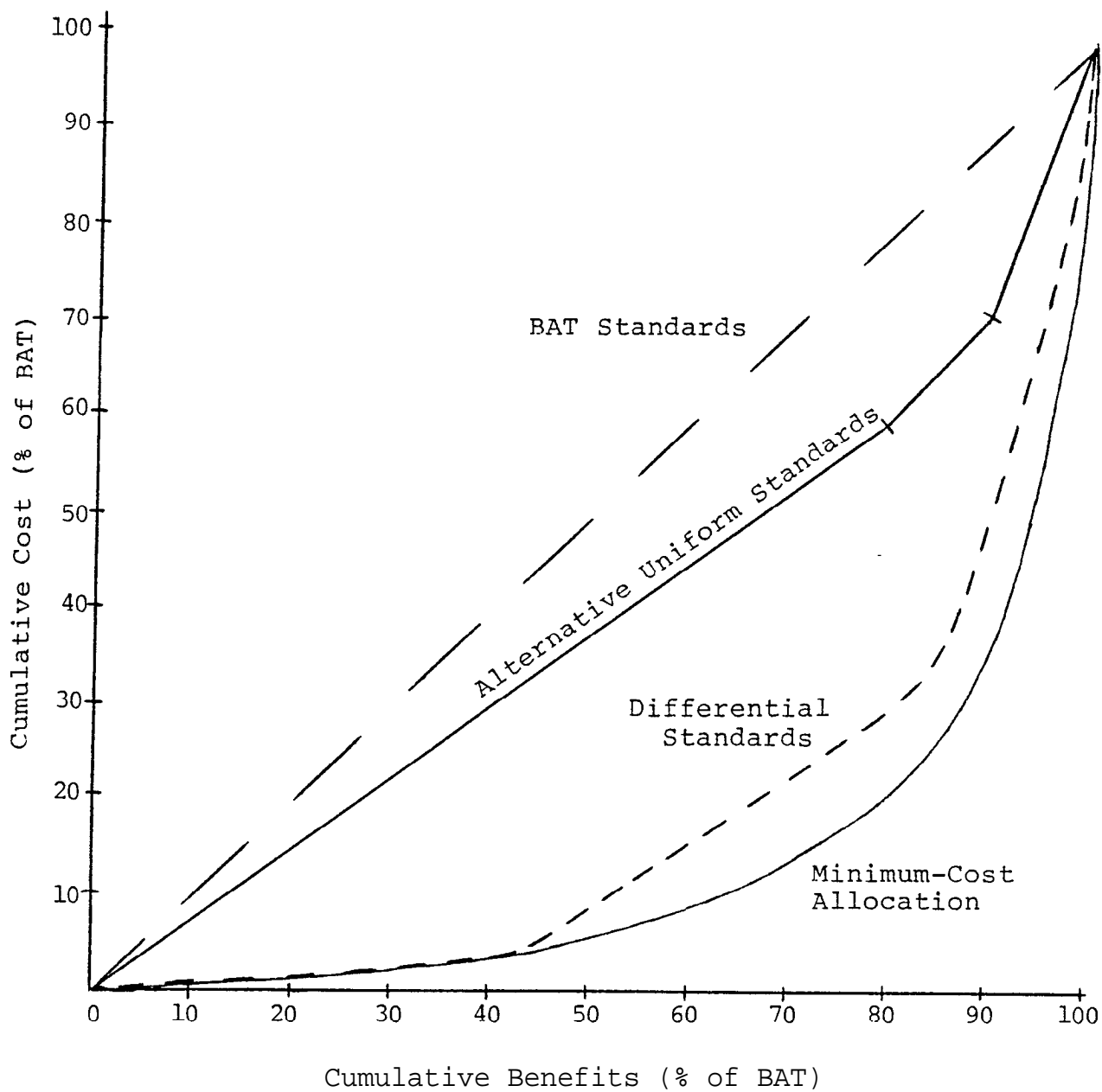
Figure 2. Cost and Benefits of Alternative Strategies for Coke Ovens

subject to the constraint that no plant could control a
particular source (e.g., doors) unless all other plants with
higher exposure factors also controlled that source. We did not,
however, require that the dividing line between "high-" and "low-
exposure plants be the same for all sources.  An efficient
combination, for example, might involve controls on doors for
plants with exposure factors of 4 $ug/m^3$-person-years/kg or
higher, controls on topside leaks for plants with exposure
factors in excess of 5, and no charging standard for any plants.
For reference, we also show the "minimum-cost" solution, which
ranks plant-source combinations in order of cost per unit of
exposure reduction, thus taking account of variations in both the
marginal costs and the marginal benefits of controlling
emissions.[13]  Note that differential standards come close to
achieving minimum cost over most of the range.  It appears that
large efficiency gains could be reaped by limiting the standards
to plants with relatively high exposure factors.

## Acrylonitrile Case Study[14]

EPA has developed data on thirty plants that might be
subject to standards limiting acrylonitrile emissions:  twelve
ABS/SAN resin plants and six in each of the other three
categories (AN monomer, acrylic fibers, and nitrile elastomer).
In all four source categories, acrylonitrile is released from
several points within plants, but production processes account
for most of the emissions (Energy and Environmental Analysis
1981).

The development of standards for acrylonitrile is at an earlier stage than for either of the other case studies. Consultant reports, however, have identified control options that we believe are representative of the BAT standards that EPA would be likely to impose if it proceeded with regulation. Table 10 lists the percentage reductions in emissions (from "uncontrolled" levels) for each source category; the control levels being considered range from 95 percent for ABS/SAN resin plants to 99 percent for firms producing AN monomer. These percentages, however, overstate the actual reductions that would be achieved given existing controls and production processes. In each category, some plants already meet the potential standard; half of the ABS/SAN resin plants use a technology that is inherently low in emissions. Moreover none of the plants is "uncontrolled." Table 10 also reports the average control levels (weighted by emissions) for plants that currently do not meet BAT standards; they range from 87 percent for AN monomer plants to 55 percent for plants producing nitrile elastomers.

For AN monomer plants, the contractors' reports provide sufficient information to adjust costs to reflect existing controls. This appears to be reasonable, as a BAT standard probably would force such plants to install controls on additional sources, rather than to replace existing controls. For the other source categories, we followed the same procedure as with maleic anhydride plants; plants were not given any cost credits for existing controls unless those controls met the

Table 10. Current Status of Plants Emitting Acrylonitrile

| Source Category | BAT Control (%) | Number of plants meeting standard | Plants requiring control | |
|---|---|---|---|---|
| | | | Number | Current Average Control (%) |
| AN monomer | 99 | 1 | 5 | 87 |
| Acrylic fibers | 95-96 | 1 | 5 | 58 |
| ABS/SAN | 95 | 6 | 6 | 70 |
| Nitrile elastomers | 96 | 2 | 4 | 55 |

Sources:

"BAT" control levels and uncontrolled emissions based on Key and Hobbs (1980) for AN monomer and Click and Moore (1979) for the other categories. Current emission estimates, used to calculate "current average control," from U.S. EPA (1983a).

likely BAT standard.  We also adjusted the estimates of both costs and benefits to reflect less-than-full-capacity operation, based on recent production levels:  95 percent utilization for AN monomer and acrylic fiber plants, 90 percent for nitirile elasatomer production, and 75 percent for ABS/SAN resin plants (Energy and Environmental Analysis 1981).

As with the other case studies, estimated exposure factors cover a wide range, from 0.009 **ug/m$^3$-person-years** per kilogram for an acrylic fiber plant to 1.14 **ug/m$^3$-person-years** per kilogram for a nitrile elastomer plant (calculated from Suta 1982b).  The average (weighted by emissions reductions) is 0.146. Thus the exposure factors for acrylonitrile appear to be substantially lower than those for benzene and far smaller than those for coke ovens.  The means for individual categories also vary widely, from a low of 0.049 for acrylic fibers to a high of 0.634 for nitrile elastomer.

**Cost-Effectiveness of BAT Standards.**  The effects of imposing BAT standards on all four source categories are summarized in the first column of Table 11. We estimate that the costs of $29 million would result in reducing exposure to acrylonitrile by just over 450,000 **ug/m$^3$**-person-years; with the CAG risk estimate, a case of cancer would be avoided roughly once every five years.  As a result, the complete set of four BAT standards clearly fails a benefit-cost test.  The cost per unitof exposure reduction is over $60 and the estimated cost per life saved is over $140 million.

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Table 11.  Cost-Effectiveness of BAT Standards for Acrylonitrile

| | | Individual Categories | | | |
| | Total | AN Monomer | Acrylic Fibers | ABS/SAN Resins | Nitrile Elast. |
|---|---|---|---|---|---|
| **Annual Costs and Benefits** | | | | | |
| Control Cost ($1000) | 28,988 | 4,792 | 6,574 | 14,317 | 3,306 |
| Reduced emissions (1000 kg) | 3,112 | 522 | 1,174 | 1,173 | 243 |
| Reduced exposure (1000 ug/m$^3$-yr) | 455 | 129 | 56 | 115 | 154 |
| **Cost-Effectiveness** | | | | | |
| Emissions ($/kg) | 9.3 | 9.2 | 5.6 | 12.2 | 13.6 |
| Exposure ($/ug/m$^3$-year) | 63.7 | 37.2 | 117.4 | 124.5 | 21.5 |
| Lives ($1 million/life) | 144 | 84 | 264 | 280 | 48 |

Sources:

    Control cost estimates based on model plant data in Key and
Hobbs (1980) and Energy and Environmental Analysis (1981) for AN
monomer and in Click and Moore (1979) for the other categories.
All costs updated to 1982 dollars using GNP implicit price
deflator.  Emissions reductions based on "current" emissions in
U.S. EPA (1983a) and on model-plant controlled emissions in Key
and Hobbs (1980) for AN monomer and in Click and Moore (1979) for
the other categories.  Exposure factors estimated using
dispersion modeling results and plant-specific population data
provided by Suta (1982b).

**Improving the Uniform Standard.** Table 11 also reports the costs and benefits of BAT standards for each of the four source categories. Note that the cost-effectiveness ratios (both for exposure and lives saved) vary widely; the ratios for ABS/SAN resin plants are almost six times higher than those for nitrile elastomer plants. Those differences reflect variations in exposure factors rather than in emission-control costs. Indeed, if we focused on cost-effectiveness in emission control rather than the more relevant measures, it would appear that ABS/SAN resin plants were better candidates for regulation than the nitrile elastomer facilities.

Restricting the BAT standards to the nitrile elastomer and the AN monomer plants would yield 62 percent of the benefits of the complete set of standards at 28 percent of the cost. The average cost per unit of exposure reduction, however, would still be over \$28 per **ug/m$^3$-person-year,** implying a cost per life saved of over \$65 million using the CAG risk estimate. Even the most cost-effective category, nitrile elastomer plants, has a cost per life saved of almost \$48 million. Thus it appears that none of the BAT standards can be justified on benefit-cost grounds.

Another possibility is to consider less stringent regulations for the individual source categories. EPA has not analyzed such alternatives, but from the model plant data it appears that a flare to control column-vent emissions from AN monomer plants would reduce emissions about 76 percent below uncontrolled levels at a cost of less than \$0.032 per kilogram of acrylonitrile (Key and Hobbs 1980, VI-5, updated to 1982

dollars).  Using the average exposure factor for those plants of 0.248 $ug/m^3$-person-years/kg, the cost per unit of exposure reduction would be $0.13; the implicit cost per life saved would be under $290,000, a relatively modest sum. All of the AN monomer plants, however, already have such flares; as reported earlier, their average current level of control is almost 90 percent.  Thus, from the data available to us, we are unable to find uniform acrylonitrile standards that are likely to yield positive net benefits given existing controls.

  **Differential Standards.**  As in the other two case studies, wide variations in exposure factors offer opportunities to improve cost-effectiveness by limiting standards to high-exposure plants.  Figure 3 plots the costs as functions of the reductions in exposure for the same strategies presented in Figure 2 for  coke ovens.  As before, the costs and benefits are expressed as percentages of those achieved under the full set of BAT standards.  Note that, as with coke ovens, differential standards based on exposure yield large savings over uniform standards and that they do almost as well as the minimum-cost allocation.  Restricting the standards to AN monomer and nitrile elastomer plants with exposure factors greater than 0.2 $ug/m^3$-person-years/kg, for example, yields almost 60 percent of the benefits of the complete set of BAT standards at only 17 percent of its cost.[8]  In contrast to the coke oven case,  however, even the differential standards fail to yield positive net benefits
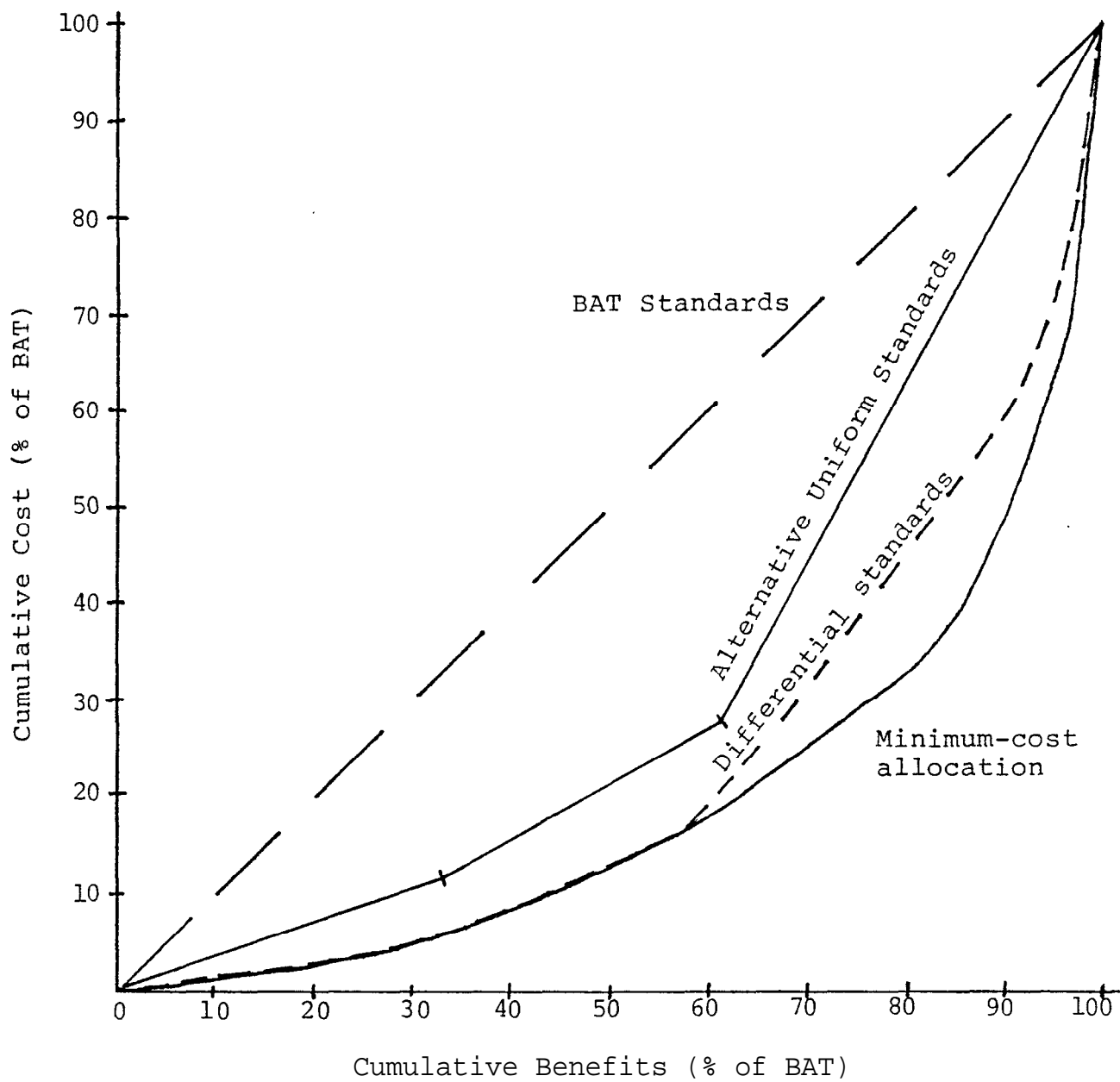
Figure 3.    Costs and Benefits of Alternative Strategies
for Acrylonitrile

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for plausible values of risk reduction. The most cost-effective plant is one in Akron, Ohio, that produces nitrile elastomer, but even its cost-effectiveness ratio is $7.98 per unit of exposure reduction, or approximately $18 million per life saved.[16] **Thus,** although differential standards can improve the cost-effectiveness ratios substantially, they appear unlikely to yield benefits commensurate with the costs of control.

## Comparisons Among the Case Studies

The three detailed case studies may be combined to provide additional insights about the use of benefit information to evaluate regulatory alternatives and establish regulatory priorities. While our overall conclusions are given in Chapter 4 after our evaluation of uncertainties, it is useful to summarize the results thus far.

**Health Benefits of Control.** Table 12 summarizes the key parameter values needed to estimate the health benefits of reducing emissions of the three substances. It shows that the value of controlling coke oven emissions will be greater than for either benzene or acrylonitrile because the unit risk factor and the average exposure factor both are much larger. On average a kilogram of coke oven emissions causes three times the exposure of a kilogram of benzene from maleic anhydride plants and over 17 times the exposure of a kilogram of acrylonitrile. The risk per unit of exposure to coke oven emissions is more than 100 times higher than for benzene and almost 30 times higher than for acrylonitrile. As a result, as shown in the last line of the

Table 12.   Risk and Exposure Information for the Three Cases

| | Maleic Anhydride (Benzene) | Coke Oven Emissions | Acrylonitrile |
|---|---|---|---|
| Unit risk factor (deaths/ug/m$^3$-yr) | $1.1 \times 10^{-7}$ | $1.3 \times 10^{-5}$ | $4.4 \times 10^{-7}$ |
| Average exposure factor (ug/m$^3$-yr/kg) | 0.721 | 2.03 | 0.146 |
| Risk per kg of emissions | $7.9 \times 10^{-8}$ | $3.7 \times 10^{-5}$ | $6.4 \times 10^{-8}$ |

table, the risk reduction from controlling a kilogram of emissions is, on average, roughly 500 times greater for coke ovens than for either of the other cases.

**BAT Controls.** Table 13 summarizes the results of our analyses of the BAT standards. (For maleic anhydride plants, the EPA has formally proposed the BAT standard, while for the other two we inferred the BAT technology from the development documents.) As expected, the health benefits are much greater for coke oven emissions than for benzene or acrylonitrile. We estimate that BAT controls on coke ovens would result in almost 11 fewer cases of cancer each year, compared to reductions of 0.4 cancer deaths for maleic anhydride controls and 0.2 cancer deaths for the four acrylonitrile standards. Though not reported in the table, our estimates suggest that controlling door leaks from a single coke plant (the one with the highest exposure factor) would generate almost three times the reduction in risk of the maleic anhydride and the four acrylonitrile standards combined (at a cost 0.6 percent as **great**).[17]

The final rows of Table 13 show equally wide differences in the cost-effectiveness of control. The final row presents the most relevant comparison -- the value placed on saving a life that is implied by the control benefits and costs. To justify acrylonitrile controls on benefit-cost grounds, the value of a statistical life would have to be at least $144 million, an implausible figure from virtually any perspective. The cost-effectiveness figure for benzene, $6.5 million, also is larger than the range of plausible estimates. Controls on coke oven

Table 13.  Benefits and Costs of Uniform BAT Standards for
          the Three Case Studies

| | Maleic Anhydride[a] | Coke Ovens | Acrylo-nitrile |
|---|---|---|---|
| **Annual Costs and Benefits** | | | |
| Control Cost ($1000) | 2,577 | 19,300 | 28,988 |
| Reduced Emissions (1000 kg) | 5,059 | 289 | 3,112 |
| Reduced Exposure (1000 $ug/m^3$-yrs) | 3,646 | 819 | 455 |
| Lives Saved | 0.4 | 10.6 | 0.2 |
| **Cost-Effectiveness** | | | |
| Emissions ($/kg) | 0.51 | 67. | 9.3 |
| Exposure ($/ug/m$^3$-yr) | 0.71 | 23.6 | 63.7 |
| Lives saved ($1 million/life) | 6.5 | 1.8 | 144. |

Note:

   [a]Based on data available to EPA when standard proposed.

emissions are the most attractive of the three BAT options, with the cost-effectiveness estimate of $1.8 million per life saved falling within the range of the published estimates. Nevertheless, all three BAT options would fail a conventional benefit-cost test based upon a value of $1 million per life saved.

Note that comparing the cost-effectiveness ratios for emission control provides a very misleading measure of the relative attractiveness of the three BAT standards. A kilogram of coke oven emissions is much more costly to control than one of either acrylonitrile or benzene. The marginal benefit of controlling coke oven emissions is so much larger, however, that coke ovens are far more cost-effective objects of regulation. This comparison provides the most compelling reason for formally evaluating the benefits of toxic control. Without considering relative carcinogenicity and relative exposure factors, it is impossible to target controls where they provide the greatest health benefits.

**Improved Standards.** We identified two strategies for improving the cost-effectiveness of controls -- relaxing the standard and focusing standards on plants located in more densely populated areas where exposure factors are relatively large. Both strategies, particularly the latter, lead to significant improvements in cost-effectiveness. Table 14 shows that such schemes reduce costs greatly with very little sacrifice in benefits. For example, relaxing the maleic anhydride standard from 97 percent to 90 percent control reduces costs to 57 percent

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Table 14.  Benefits and Costs of Alternatives as Percentages
           of BAT Levels

| | Maleic Anhydride[a] | Coke Ovens | Acrylonitrile |
|---|---|---|---|
| Relaxed Uniform Standard[b] | | | |
| Benefits | 94 | 80 | 62 |
| Costs | 57 | 61 | 29 |
| Differential Standard[c] | | | |
| Benefits | 96 | 81 | 60 |
| Costs | 37 | 33 | 18 |

Notes:

[a]Based on data available to EPA when standard proposed.

[b]Defined as:
  maleic anhydride:  90 percent
  coke ovens:  doors only
  acrylonitrile:  AN monomer and nitrile elastomer plants

[c]Defined as:
  maleic anhydride:  97 percent control for plants with exposure
    factors greater than 0.6
  coke ovens:  doors and topside for plants with factors greater
    than 2.0
  acrylonitrile:  BAT controls for AN monomer and nitrile elastomer
    plants with exposure factors greater than 0.2

of the BAT level while retaining 94 percent of the benefits.  The differential standard performs even better in each of the case studies; for maleic anhydride, focusing the BAT standard on the four plants with the greatest exposure factors yields 96 percent of the benefits at only 37 percent of the costs.

These modifications to the standards reduce costs much more than they reduce benefits, but most do not result in cost-effectiveness ratios within the range of published estimates, as shown in Table 15.  That table reports both the cost-effectiveness ratio for each of the alternative strategies and the incremental ratio for switching from that alternative to BAT. For acrylontrile, even the differential standard results in an implicit value per life saved of $42 million.  (Note that the cost-effectiveness of the BAT standard jumps to $286 million when incremental costs and benefits are evaluated.)  The benzene alternatives yield estimates as low as $2.5 million per life saved.  The coke oven alternatives result in the lowest figures, $1.4 million for the improved standard and $730,000 for the differential  standards.

**Benefit-Cost Comparisons.**  The final step in a full benefit-cost evaluation is to compare the net benefits of the regulatory alternatives, both across pollutants and across alternatives. Table 16 provides our estimates based upon a value of $1 million per life saved.  Perhaps the most striking result is that the differential standard for coke oven emissions is the only alternative with positive net benefits (which, of course, was implied by the cost-effectiveness results).  The other

Table 15.  Cost per Life Saved (in $1 million) of Alternatives
           to BAT for the Three Case Studies

| | Maleic Anhydride[a] | Coke Ovens | Acrylonitrile |
|---|---|---|---|
| Relaxed Uniform[b] | 3.9 | 1.4 | 64.2 |
| incremental BAT | 41.6 | 3.7 | 274. |
| Differential[c] | 2.5 | 0.73 | 42.1 |
| incremental BAT | 80.4 | 6.5 | 286. |

Notes:

   See corresponding notes in Table 2.14.

alternatives result in net losses ranging from $0.6 million for
differential standards for maleic anhydride plants to $28.8
million for the BAT standards for acrylonitrile plants.

Note that the "improved" standard for coke oven emissions
results in a larger net loss than the much less cost-effective
BAT standard for maleic anhydride plants.  This somewhat
paradoxical result simply reflects the larger costs of the coke
oven standards.  Also note that the differences in net losses
among the alternatives for acrylonitrile reflect almost entirely
differences in control costs, as the benefits are trivial. The
net gain in shifting from the full set of BAT standards to the
relaxed uniform standard, for example, consists of a cost
reduction of $20.1 million and a benefit reduction of just
$76,000.

The net benefit (loss) estimates in Table 16 do not
reflect optimization of the various regulatory alternative.  For
a value per life saved of $1 million, the optima for
acrylonitrile and for benzene emissions from maleic anhydride
plants appear to be no additional controls, which would yield
zero net benefits (by definition).  For coke ovens, the optimal
uniform standard is also zero additional controls.  The optimal
differential standard is less stringent than the alternative
represented in Table 16; for a value per life saved of $1
million, maximum annual net benefits of $3.7 million are achieved
when door and topside controls are required only for the four
coke plants with exposure factors greater than 5 $ug/m^3$-person-
years per kilogram.[18]

Table 16.   Net Benefits (million $/year) of Alternative Strategies
           for a Value per Life Saved of $1 million

|  | Maleic Anhydride | Coke Ovens | Acrylonitrile |
|---|---|---|---|
| BAT | -2.2 | -8.7 | -28.8 |
| Relaxed uniform | -1.1 | -3.2 | -8.0 |
| Differential | -0.6 | 2.3 | -4.9 |

**Summary.** These detailed results indicate that uniform technology-based controls will have vastly different net benefits depending upon the pollutant and the source category; the implicit cost per life saved of BAT standards varies by more than a factor of 100 even within our limited sample. Moreover, in each of the three cases we can identify alternative standards that would yield higher net benefits than BAT for any plausible value of risk reduction. For two of the three cases, however, even the most cost-effective standards available appear to fail any reasonable benefit-cost test. In the third case, coke oven emissions, only when the standard is relaxed and restricted to high-exposure plants does regulation appear to yield positive net benefits for a value per life saved of $1 million.

These conclusions must be viewed as tentative, for we have not yet taken account of the substantial uncertainties associated with the estimates, particularly for the benefit estimates that play an important part in our recommendations. We now turn to an examination of those uncertainties to determine how robust the conclusions are likely to be.

### III.  UNCERTAINTIES IN THE BENEFIT ESTIMATES

The benefit estimates in Chapter 2 are based primarily on information developed by EPA.  With the exception of the value of saving a life, we have used point estimates for each of the relevant parameters.  Most of the parameter values, however, are highly uncertain.  The most important and well-known of these uncertainties concern the unit risk estimates (where the plausible range covers several orders of magnitude or more), but many of the other parameter estimates also are subject to considerable uncertainty and dispute.

In this chapter, we consider how these uncertainties affect our estimates and conclusions.  The key question is not whether the estimates are precise -- for we freely concede that they are not -- but rather how robust are our conclusions in the face of substantial uncertainties and potential errors.  Consideration of the uncertainties and their impacts on the conclusions also suggests where it might be most profitable to devote resources to reduce the range of uncertainty.

Our discussion focuses in turn on each of the four steps in benefit estimation shown in Figure 1.  We begin with the largest uncertainty, the estimate of unit risk, and then turn to the other three -- emissions, exposure and the valuation of risk reduction.  In each case we deal both with the generic problems and with specific examples that arise in the case studies.  Our main conclusion is that while the benefit estimates used in Chapter 2 are very imprecise, in most cases they appear to be

biased upwards.  Thus consideration of the uncertainties reinforces our tentative conclusion that in none of the three cases is a uniform BAT standard justified.

## Unit-Risk Estimates

In each of the three cases, epidemiological evidence of carcinogenicity provides the primary argument for listing the substance as a hazardous air pollutant under Section 112.  This is in contrast to many suspect chemicals where the only evidence comes from experiments with laboratory animals, usually mice or rats.  Thus, in none of the cases do we face the difficult and controversial task of extrapolating from animals to humans (Crouch and Wilson 1982, 64-68).  Predicting low-dose risks, however, remains highly uncertain because the epidemiological studies all involved workers exposed to far higher concentrations of the substances than are members of the general population (or even workers under current conditions).

**Low-Dose Extrapolation.**  The problem of extrapolating from high-dose data to low-dose exposures is ubiquitous in the regulation of environmental carcinogens.  The central problem is that neither epidemiological studies nor laboratory experiments with animals are capable of detecting low-level risks.  Thus, unless the chemical is a very potent carcinogen or the type of cancer caused is ordinarily very rare, individuals with unusually high exposures must be studies, or animals must be given doses far beyond those ever likely to be encountered by people. A variety of mathematical models has been developed to perform the

necessary extrapolations.  Unfortunately current theory does not provide unambiguous support for any one of them, nor can they be selected empirically.

The "one-hit" model is the one most often used.  It assumes, at least metaphorically, that cancer can be induced by a single "hit" of a susceptible cell by a carcinogen. Thus, the lifetime risk is the probability of one or more hits. At low exposure levels, the predicted risk is proportional to the dose; thus, for example, if the relevant dose is 1000 times lower than that at which the risk was measured, the estimated risk is also 1000 times lower.  Because of this property, it is often called the "linear" model.  It is difficult to tell how much of this model's popularity is due to scientific belief in its accuracy as opposed to a value judgement that decision makers should be conservative in the face of great uncertainty; most scientists accept the linear model as providing an upper-bound estimate of the risk.[19]

The other models commonly used all are convex at low doses; as the dose is reduced, risk falls more than proportionately. Thus, when estimated from the same data, these models predict smaller low-dose risks than the linear model.  The most well-known of these nonlinear models is the Mantel-Bryan (1961) procedure, which is a special case of the log-probit model. Other models include the logit, which yields an S-shaped curve similar to the log-probit (but with somewhat higher risks at low doses) and the multi-hit model, which is a generalization of the one-hit model.

All of the models yield similar dose-response curves over the ranges that can be measured in laboratory and epidemiological studies. When extrapolated to the low concentrations relevant to most EPA decisions, however, the models predict radically different risk levels.[20] Indeed, when the extrapolation covers two or more orders of magnitude, as is typically the case when occupational epidemiological studies are used to predict risks from ambient concentrations, for all practical purposes the nonlinear models' estimates may be treated as zero because they are so much lower than the linear model's predicted risks. Thus, with rare exceptions for extremely potent carcinogens, regulations to reduce exposure of the general population to environmental carcinogens must rest on a belief that the linear model has a nontrivial probability of being correct (or at least is a good approximation of the true dose-response model).

From a decision analytic perspective, the ideal approach would be to assess the probabilities that each of the models is correct, and then to use those probabilities to compute an expected dose-response function. Unfortunately, we are not aware of any attempt to develop such probabilities based on expert opinion. We do feel confident, however, that an expected dose-response function would be approximately linear at low doses, not because we are certain that the linear model is correct, but rather because the other models predict such tiny risks that the linear estimate would dominate so long as even a small probability was assigned to the linear model. Note also that the unit risk factor for this expected dose-response model would not be as large as that estimated by the linear model alone; it would

be approximately the pure linear estimate times the probability assessed that the linear model is correct.

This line of argument suggests that while it is appropriate to assume that the expected benefits of control are proportional to the reduction in exposure, our estimates of reduced mortality probably are too high, perhaps by a substantial margin, because they rely on CAG unit-risk estimates, which implicitly assign a probability of unity to the linear model's being correct.

**Uncertainties in Applying the Linear Model.** In addition to disputes about the appropriate model for low-dose extrapolation, estimates of the unit risk factors are plagued by uncertainties about how to interpret the epidemiological data. One major difficulty is that the exposure levels for the individuals in the epidemiological studies often are extremely uncertain; typically exposures occurred over many years when few, if any, measurements were made of concentrations. Many other problems, however, also can cloud the use of epidemiological data to establish a base from which to extrapolate.

The controversy surrounding the CAG's risk estimate for benzene illustrates many of these issues. The record for benzene is more complete than that for the other substances because it is the only one that actually has been listed by EPA and because it was the subject of an important Supreme Court case regarding OSHA's attempt to tighten the occupational standard.

The CAG (Albert et al, 1979) based its unit risk estimate on data from three epidemiological studies: one of workers in two plants using benzene as a solvent to make a transparent film

(Infante et al 1977), another of Turkish shoe workers using benzene-based adhesives (Aksoy et al 1974 and 1976 and Aksoy 1977), and the third of workers in chemical plants using benzene (Ott et al 1977). In applying the linear model to each of these studies, the CAG had to make several assumptions, some of which have been criticized severely. The issues raised have included the CAG's exposure estimates for all three studies, its inclusion of the deaths of two workers not in the original cohort of the Infante study, its failure to exclude workers exposed to other hazardous chemicals in the Ott study, and its estimate of the baseline risk in the Aksoy study (see Nichols 1981, ch. 9, for a more detailed summary of these criticisms). Two EPA analysts, Luken and Miller (1979), concluded that the CAG risk estimate was too high by a factor of four. Lamm (1980), an occupational physician who testified for the American Petroleum Institute at hearings on the proposed standard for maleic anhydride plants, argued that the CAG estimate should have been lower by more than a factor of ten. These differences are all the more startling because they were based on the same studies and the same model as the CAG estimate.

**Noncarcinogenic Effects.** One possible source of downward bias in our benefit estimates is that we have looked only at reductions in cancer risks. Each of the substances also has been associated with other adverse health effects at relatively high doses. Very high exposures to benzene or acrylonitrile rapidly lead to death. Chronic, high-level occupational exposures to benzene also have been associated with increased risks of

aplastic anemia and other serious blood disorders (U.S. EPA 1978). In contrast to carcinogenic effects, however, most scientists accept the concept of thresholds for noncarciogenic hazards, and current environmental exposures appear to be far below the relevant levels.

Chromosomal damage may be of potential concern at low doses. All three substances appear to cause such damage, based on evidence from human studies or short-term mutagenicity tests. None of the substances, however, has been associated with birth defects, and the data are insufficient to derive even crude dose-response relationships. Moreover, the relevant EPA documents emphasize mutagenic effects as corroborating the carcinogenicity of the three substances, rather than as seperate concerns. Thus we do not believe that we have neglected significant health benefits by dealing exclusively with carcinogenic effects. We also note that for benzene and acrylonitrile, particularly the latter, the non-cancer benefits would have to be substantially larger than the cancer benefits to justify the BAT standards on benefit-cost grounds.

 **Summary.** Disputes about the appropriate dose-response model and about how to interpret highly imperfect epidemiological studies mean that it is impossible to develop unit risk estimates for any of the three substances that can be defended rigorously. We do not claim to be experts in risk assessment. It appears, however, that the unit-risk estimates used in Chapter 2 are biased upwards, primarily because they are based solely on the linear model. In addition, at least in the case of benzene, we

have reason to believe that the CAG has followed procedures that lead to an overestimate of the linear model's coefficient. The exclusion of non-cancer benefits introduces a potential source of bias in the opposite direction, but we do not have any evidence to suggest it is large relative to the debate over the carcinogenic effects. To the extent that the unit risk factors are too high, we have overestimated the expected benefits of all of the strategies for all of the cases. Revising those estimates downwards reinforces our conclusions regarding benzene and acrylonitrile. It also reinforces our conclusion that uniform BAT standards on all three sources of emissions from coke oven plants would not be cost-effective relative to less stringent regulations (or none at all).

## Uncertainties in Emissions

The change in emissions due to regulation is perhaps the most straightforward of the calculations that produce benefit estimates. For each plant, EPA estimates the current (or baseline) emissions and then estimates the emissions with controls in place. The difference between these two estimates yields the reduction in emissions that becomes the input into the meteorological model used to predict concentrations in the next step of the process. Despite this apparent simplicity, however, estimates of the reduction in emissions are far from precise. Indeed, as discussed below, emissions may be the largest source of uncertainty in estimating the benefits of regulating coke

plants.  We discuss first the general issues and then the particular problems associated with the coke-oven estimates.

**General Uncertainties.**  Several uncertainties are common to emission estimates for all three cases, and indeed to the vast majority of regulations likely to be considered under Section 112.  These uncertainties are particularly great, on a proportional basis, at the level of individual plants.

The first problem is that emission estimates are based on a model plant and then projected to actual individual sources using a limited number of plant-specific factors.  In the case of benezene emissions from maleic anhydride plants, for example, EPA's emission estimates assume that all plants achieve a 94.5 percent conversion rate, although a contractor has estimated that conversion rates vary across plants from 97 down to 90 percent (U.S. EPA 1980, 1-7).  This range may appear small, but it is important given that the uncontrolled emission rate from a plant is proportional to the difference between its conversion rate and 100 percent.  Thus, a plant with a 90 percent conversion rate has more than three times the uncontrolled emission rate of a plant that achieves 97 percent conversion.  Similar problems affect plant-specific estimates for nitrile elastomer plants, where conversion rates for the acrylonitrile monomer vary from 60 to 90 percent, depending on the particular type of nitrile rubber produced (Radian Corporation 1982, 43).

Estimates of emissions reductions also are complicated by uncertainty about the effectiveness of existing controls, if any.  In all three cases, most of the plants already have emission

controls of some kind, due to state regulations, OSHA standards, or economic self interest in recovering valuable feedstock or byproducts. EPA has tried to ascertain the effectiveness of such controls, but in most cases the estimates are crude, based on nominal capabilities rather than actual monitoring.

A final general problem in estimating emission reductions concerns production levels. Emissions are a function of both emission rates and the level of capacity utilization. Our estimates in Chapter 2 for the acrylonitrile plants have been adjusted for recent production levels, but the estimates for maleic anhydride plants and coke ovens assume full-capacity operation. Few plants, however, operate at full capacity, so our benefit estimates are too high. This poses a severe problem where control techniques are capital-intensive, because most of the costs are fixed while the benefits vary with production levels. Nichols (1981), for example, shows that the cost per unit of benefit almost doubles for maleic anhydride plants if they operate at 56 percent of capacity (the average in 1977) rather than 100 percent. We suspect that the cost-effectiveness ratios for coke ovens would be affected much less drastically, however, because most of the costs are ongoing maintenance expenses and thus should vary with production levels.

Even if emission estimates are accurate at the time they are made, they may not provide reliable projections of the impact of a proposed regulation. The effects are most dramatic in the case of maleic anhydride plants where, as reported in the previous chapter, all of the uncontrolled plants identified by EPA in 1980 when the regulation was proposed have since closed, switched

feedstocks, or installed controls.  In the case of coke ovens,
given the depressed state of the steel industry, it is quite
possible that additional plants might close over the next few
years.  (It is also possible, of course, though we suspect less
likely, that as the recovery continues some shut-down plants may
be reopened.)

The issues discussed above contribute to the uncertainty
surrounding the emission estimates, but, so far as we can tell,
do not point to any clear bias in the estimates we used in
Chapter 2.  (The obvious exception is maleic anhydride plants,
where we know that conditions have changed dramatically.  There,
however, we have taken the perspective of a regulator making a
decision at the time the standard was proposed.)

**Coke Ovens.**  By far the largest uncertainties about
emissions reductions arise with regard to coke ovens.  EPA's
contractor presents "minimum" and "maximum" estimates for
emissions (both baseline and post regulation) for all three
emission sources.  The ratios of maximum to minimum estimates are
11.2 for doors and 6.4 for topside leaks.  The ratios for
charging vary across plants from a high of over 650 to a low of
360 (calculated from U.S. EPA 1983b).  These wide disparities for
all three sources appear to reflect two factors.  The first is
that the emissions affected by the potential regulations are
"fugitive" emissions, rather than emissions released from normal
process operations, so they are highly dependent on source-
specific conditions and practices.  According to EPA (1981b), the
emission rate for doors is dependent on the time into the coking

cycle, the gap size of the metal-to-metal seal, and the oven temperature and pressure. The emissions rates for lids and offtakes are dependent on worker practices in applying luting mixtures, on pressure fluctuations in the oven, and on the gap size of the emission point. Charging emission rates are a function of the time of the charge, pressure fluctuations, and gap size around the drop sleeves and the charging ports. In addition, all of the emission rates are affected by the type of coal used, which can vary not only from plant to plant but from day to day for a particular plant.

The second important source of uncertainty is also related to the fact that the emissions are fugitives. Because of the difficulty of measuring such emissions (there is no single stack to monitor), the regulations being considered by EPA are stated in terms of visible emissions, rather than as a limit on mass emissions. The charging standard under consideration, for example, sets an upper bound on the number of seconds of visible emissions during the charging cycle. The agency is very uncertain about the relationship between visible emissions and mass emissions (the relevant measure for predicting benefits).

Our estimates in Chapter 2 represent a simple average of the minimum and maximum estimates for each plant. Table 17 shows the effects on the exposure cost-effectiveness ratios of using either the minimum or maximum estimates. With the maximum emission estimates, the cost-effectiveness of the BAT standard falls from $23.6 to $12.8 per $ug/m^3$-person-year ($1.0 million per life with the CAG risk estimate). Note that even with the

Table 17.  Cost-Effectiveness Ratios for Alternative
Coke-Oven Emission Estimates

| Emission Estimate | Control Options | | | |
|---|---|---|---|---|
| | All | Doors | Topside | Charging |
| Average | 23.6 | 17.8 | 23.4 | 71.3 |
| Maximum | 12.8 | 9.7 | 13.5 | 38.7 |
| Minimum | 146. | 109. | 86.5 | 21,700. |
| Mean (log normal) | 37.8 | 27.7 | 31.2 | 266. |

Note:  All entries are in $/ug/m$^3$-person-years.

maximum estimates, charging is not cost effective unless the value of exposure reduction is at least $39 (about $3 million per life saved).  The effects of substituting the minimum emission estimates are much more dramatic.  The cost-effectiveness of the overall standard falls to $146 per $ug/m^3$-person-year (over $11 million per life), and to justify even the door standard requires a  value of over $100 per unit of exposure reduction (over $8 million per life).

EPA's documentation does not assess the reliability of either the minimum or maximum estimates.  We were unable to elicit a "best" estimate, and thus relied on a simple average of the two.  It is unlikely, however, that estimates with such wide variation follow a symmetric distribution (such as the normal distribution), and thus an average of the "high" and "low" estimates is unlikely to be a good estimate of the mean.  As an alternative, we assumed that the emission estimates were distributed log normally and that the minimum and maximum estimates represented the 95th percentile confidence limits of the distributions.  With those assumptions, it is possible to estimate the variance of the log of emissions for each source, and from that to estimate the expected value of emissions. In every case, the mean calculated in this manner is lower than the simple average of the minimum and maximum emissions.  For doors, the mean is about 67 percent of the estimate in Chapter 2.[21]  For topside leaks, the percentage is 76.  For charging, where the maximum/minimum ratio varies across plants, the difference between the mean and the Chapter 2 estimates also varies across plants; on average the mean is lower by almost a factor of 3.5.

The last line of table 17 shows the cost-effectiveness ratios using these alternative emission estimates.  Note that the cost-effectiveness ratio for the BAT standard ("all") rises to over $37 per $ug/m^3$-person-year (almost $3 million per life with the CAG risk estimate).

**Summary.**  Uncertainties about emissions appear to be potentially important only in the case of coke ovens.  That reflects two facts:  (1) the uncertainties are much larger for coke ovens than for either of the other cases and (2) the coke oven decision is the "closest" one, with cost-effectiveness ratios in the plausible range.  Even with the maximum emission estimates, however, it is not clear that the uniform BAT standard yields positive net benefits.

Our results suggest that it would be useful to try to narrow the range of estimates of emissions from coke ovens. This would appear to be especially critical if the tentative decision were to proceed with regulation, as a plausible benefit-cost case for the BAT standard is possible only if actual emissions are in the upper end of the estimated range.  If it is not possible to reduce the range significantly, a better attempt should be made to assess the uncertainties and to derive a careful estimate of the expected value of emissions reductions.  As our calculations with a log-normal distribution indicate, the mean may be substantially lower than a simple average of the "minimum" and "maximum" estimates.

## Uncertainties in Exposure

In all three cases, as discussed in the previous chapter, the exposure factors for each plant were based on generalized dispersion modeling and plant-specific population data. Several types of uncertainty affect the accuracy of the exposure estimates: (1) general questions about the accuracy of dispersion models, in particular their reliability at substantial distances and their ability to predict concentrations indoors, where individuals spend most of their time; (2) the applicability of general dispersion modeling to individual plants; and (3) differences between residential population densities (used to estimate exposures) and time-weighted densities that account for time spent away from home. We deal with each of these issues in turn, providing examples from the individual cases as appropriate.

**General Accuracy.** Dispersion models for toxic air pollutants typically are quite simple compared to models for water or ground pollutants, or for air pollutants for which atmospheric chemical interactions and long-range transport are important. The meteorological inputs usually include wind speed, direction, and turbulence. The models also require inputs specifying the characteristics of the source, such as the height and velocity of releases. The accuracy of these dispersion models is uncertain; calibration is difficult because in many cases it is hard to relate measured concentrations to the individual sources modeled (Miller 1978). In the case of acrylonitrile, Suta (1979) compared dispersion modeling estimates

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with monitoring data for eight plants.  The model estimates were about 30 percent higher than the actual measurements, but Suta concluded that the fit was quite close because the monitoring method used tends to understate actual concentrations by roughly the same margin.

The accuracy of the models deteriorates as the distance from the source increases.  As a result, dispersion modeling for cases such as these usually is not carried out beyond 30 km, the maximum distance used for both coke ovens and the acrylonitrile plants.  In theory this truncation introduces a bias, understating total exposure levels.  Concentrations at greater distances, however, typically are very low.

The modeling for benzene from maleic anhydride plants was carried out only to 20 km, which raises the concern that our comparisons across the case studies may be distorted by arbitrary differences in the scope of the exposure assessments.  To check for that bias, we reestimated exposures for coke ovens and acrylonitrile using data out to only 20 km.  The results were reassuring:  total reductions in exposure fell by only 9 percent for coke ovens and by 11 percent for the acrylonitrile plants. Thus it does not appear that the difference in distances has a significant impact on the relative rankings of the three case studies.

Additional uncertainty is introduced by the fact that the models are designed to predict outdoor concentrations, but most people spend the vast majority of their time indoors. Recent studies of "indoor air pollution" suggest that concentrations of pollutants indoors may be very different than those outdoors.

Many of these studies, however, have involved pollutants for which there are sources indoors as well as outdoors. Theory and monitoring data both indicate that for pollutants without indoor sources, average concentrations indoors will be the same as or lower than those outdoors (Spengler and Sexton 1983). Thus it appears that, to the extent that the use of outdoor concentrations to estimate exposure levels introduces any bias, it is in the direction of overstating the benefits of the regulations.

**Plant-Specific Modeling.** In none of the three cases were plant-specific data used to calculate exposure factors. The coke oven (U.S. EPA 1981b) and maleic anhydride (U.S. EPA 1980) analyses both used Pittsburgh meteorological data for all plants. The acrylonitrile results are based on generalized conditions rather than actual data from any particular area (Suta 1979). The failure to use plant-specific data clearly increases uncertainty about the exposure estimates for individual plants. Exposure levels around a particular plant, for example, will depend critically on whether prevailing winds blow toward or away from densely populated areas. It is not clear, however, how it affects the overall estimates.

The evidence that we have available on this issue is limited and mixed. In the background document for maleic anhydride plants, EPA states that "meteorological conditions that maximize ground level concentrations... are common in the Pittsburgh area" (EPA 1980, 4-11). In the supporting documents for coke ovens, however, the agency reports that it also tried data from Chicago

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and Birmingham; the Chicago results were roughly the same as those for Pittsburgh, but the Birmingham data generated concentrations that were two to three times higher (U.S. EPA 1981, app. E).

Dispersion modelling performed for maleic anhydride plants by the Chemical Manufacturers Association (CMA) suggests that plant-specific parameter values may be important. The CMA used a model similar to the proprietary model used by EPA's contractor. (The proprietary model reportedly is a modified version of the CRYSTER model used by CMA.) The CMA used EPA's population data, but plant-specific data on such parameters as stack height, exit velocity, and gas temperature. It also used the closest available meteorlogical data for each plant (Galluzzo and Glassman 1980). The exposure factors calculated from the CMA results for individual plants ranged from 62 percent lower to 16 percent higher than those derived from the EPA modeling with uniform parameters. On average, the CMA results were 37 percent lower (Nichols 1981, 333).

We remain uncertain as to whether the use of general parameters in the dispersion modeling introduces a systematic bias in the overall results. It does appear clear, however, that greater accuracy could be achieved through the use of more plant-specific parameters. It would seem particularly easy to use local meteorological data.

**Population Patterns.** The final uncertainty in the exposure factors arises because the EPA estimates implicitly assume that individuals spend all of their time close to their homes; the

population data are based on place of residence.  This probably

creates little problem for children, who are likely to attend

nearby schools, or for non-working adults who spend most of their

time at home or visiting friends or stores closeby.  It may,

however, create larger inaccuracies for adults who work at sites

far away; to the extent that concentrations where they work are

different than those at home, the exposure factors will be

inaccurate.  Individuals who live closer to sources than they

work will face lower exposures, while those who work closer to

sources will experience higher than predicted exposures.  We do

not have the data to estimate the empirical magnitude of this

uncertainty for the case studies, but believe that it is unlikely

to be significant compared to the other sources of uncertainty.

   **Summary.**  The data available do not allow us to quantify the

uncertainty about the exposure factors.  The uncertainties are

greatest at the level of individual plants, in part because of

the failure to use plant-specific values for any parameters other

than population.  The overall estimates should be more accurate,

if only because many of the plant-specific errors are likely to

cancel out each other.  Unlike some of the other steps in benefit

estimation, we cannot identify any significant sources of bias.


## Reconsidering the Case Studies

   As the discussion in this chapter has made clear, huge

uncertainties pervade estimates of the benefits of regulating

airborne carcinogens.  As a result, the figures that we presented

in Chapter 2 must be viewed with a very strong dose of

skepticism; they may well be in error by orders of magnitude. Despite the imprecision of the numerical estimates, we believe that the issues raised in this chapter reinforce most of our earlier conclusions.  More specifically, to the extent that we can identify significant likely biases in those estimates, they are in the direction of having overestimated the expected benefits of regulation.

The conclusions are clearest for the four source categories emitting acrylonitrile and for maleic anhydride plants emitting benzene.  For acrylonitrile, the cost-effectiveness ratios were an order of magnitude or more higher than the plausible range of values of risk reduction.  Nothing in this chapter has suggested that out estimates are in error by that margin.  (Unless, of course, one favors one of the nonlinear dose-response models, but that would cut in the other direction.)

The results in Chapter 2 for maleic anhydride plants were substantially closer, at least using the data available to EPA when it proposed the standard, though the estimated cost per life saved was still in excess of $6 million.  Several factors raised in this chapter suggest that a more accurate estimate of the expected cost-effectivness ratio would be substantially higher. These include:  (1) the general issue of the appropriate dose-response model; (2) evidence that the CAG overestimated the linear model's risk factor; and (3) a significant rise in the cost per life saved when the estimates are adjusted for less than full capacity operation.  The recent developments summarized in Chapter 2 -- plant closures, conversions to n-butane, and the

installation of controls at Monsanto -- add further weight to the argument that the proposed regulation would provide minimal benefits.

Our results in Chapter 2 were most ambiguous for coke ovens, although it appeared that a BAT standard for charging emissions almost certainly would fail a benefit-cost test. The information provided in this chapter reinforces that conclusion; even with the "maximum" emission estimates, the charging standard fails to yield positive net benefits for plausible values of risk reduction.

Whether the uniform door and topside standards generate positive expected net benefits remains in doubt. Several issues raised in this chapter, however, tend to argue against those standards: (1) the likelihood that the pure linear model overestimates the expected risk; (2) the examples suggesting that it is difficult to justify placing a value on risk reduction much in excess of $1 million per expected life saved; and (3) the asymmetry in uncertainty about emissions, with some basis for believing that the expected levels of emissions reductions are lower than those we used in Chapter 2 based on a simple average of the maximum and minimum estimates.

These same issues raise questions about whether even differential standards limited to high-exposure coke plants are likely to yield positive net benefits, though it is clear that they are more efficient than the uniform standards. Although many of the uncertainties are unlikely to be resolved, it should be possible to improve the estimates of emissions reductions and exposure factors. Note that if a differential strategy were to

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be followed, it would not be necessary to investigate these issues for all coke plants, but rather only for those that are located in areas with relatively dense populations.

## Uncertainties in Valuing Risk Reduction

Most popular criticisms of the application of benefit-cost analysis to regulations designed to reduce risk focus on the difficulty of assigning a "value to life." In contrast to the other uncertainties we have discussed, the problem of valuing risk reduction is not one of "science." Ultimately the tradeoff between greater protection and higher costs must be made by responsible public officials, although studies by economists and others can help inform the debate.

The empirical studies of willingness to pay for risk reduction cited in Chapter 2 cover a wide range, roughly an order of magnitude, from several hundred thousand to several million dollars per life saved. As discussed in that chapter, however, even that wide range is sufficient to reject the BAT standards for maleic anhydride plants and for all four types of plants that emit acrylonitrile. It is also sufficient to indicate cost-beneficial modifications of the coke-oven regulations, though not precise enough to determine if more limited regulations of coke ovens are justified. Thus it appears that uncertainty about the value of risk reduction does not pose the insurmountable obstacle to benefit-cost analysis that many have claimed.

**Narrowing the Range.**   Some examples may make it easier for many readers to narrow the range.   The first example is due to Bailey (1980) who considers a hypothetical program that lowers the annual risk of death by 0.0005 (roughly the decline in U.S. death rates from 1970 to 1975).   He asks how much a family of four with an income of $18,500 (about the median in 1978, the year for which he made his estimates) would be willing to pay for such a program, which "saves" 4(0.0005) = 0.002 lives per year for the family.   Using Bailey's intermediate estimate of $360,000 per life, the family would be willing to spend up to $720, about 4 percent of its income.   If the value per life saved is L = $1 million, it would be willing to spend $2000, roughly 11 percent of its income.   Both estimates strike us as plausible.   If the value per life saved is $2 million, however, the family would be willing to spend $4000 per year; we find it difficult to believe that many families would be willing to sacrifice over one-fifth of their income to face the death rates of 1975 rather than those that prevailed in 1970.   If we use the very high end of the range, with L = $5 million,the family's willingness to pay rises to $10,000, over half its income.

Consider another example, a hypothetical new automobile technology that cuts in half the risk of a fatal accident.   (To keep matters simple, we make the unrealistic assumption that it has no impact on nonfatal injuries; including nonfatal injuries would increase willingness to pay.)   As there are roughly 50,000 automobile-related fatalities each year, such a technology would save 25,000 lives annually.   If we value each life saved at L, we

should be willing to spend up to 25,000L annually to use this technology; e.g., if L = $1 million it is worth $25 billion, and if L = $3 million we should be willing to spend up to $75 billion per year as a nation. These costs may be easier to grasp if we convert them to a cost per new car; with roughly 10 million new cars sold each year, L = $1 million implies that an individual would be willing to pay up to $2,500 extra to buy a car with this technology. We suspect that some readers would accept this option, though many would not. The implications of higher values, however, are much less plausible; if L = $3 million, for example, new-car buyers should be willing to pay up to $7,500 extra to purchase an automobile with these safety features.

Some readers may object to this example on the grounds that while automobiles represent a voluntary risk (at least for the owner), where the same person bears the costs and the risks, environmental carcinogens impose involuntary, concentrated risks on relatively small groups of individuals, and thus society should be willing to spend much more to control them. We shall not attempt to deal with this argument in detail, but at least three factors suggest that it is less compelling than it may appear at first. (1) The levels of risk imposed by airborne carcinogens, even for nearby residents, typically are very small relative to many other risks that individuals run routinely; we are not dealing with cases where identified individuals face unconscionably high risks.[22] (2) Although the risks associated with particular types of sources (such as maleic anhydride plants) often are borne by relatively small numbers of people, the overall risk from environmental hazards is distributed much

more evenly.  Most exposure to benzene, for example, is caused by automobiles and service stations (Mara and Lee 1978).  (3) Regulations impose involuntary costs at the same time they reduce involuntary risks.  Few of these costs are borne by the owners of the firms regulated; most are passed on to consumers of a wide range of products.  For example, the cost of controlling coke oven emissions would affect the price of steel and hence the prices of goods that have steel components. The cost of controlling acrylonitrile would affect, among other items, the prices of many types of clothing. As a result, overall there is tremendous overlap between those who pay for tighter environmental controls and those who benefit from them.

**Years of Life and Discounting.**  Two factors suggest that we might wish to ascribe a lower value to "lives saved" through the regulation of environmental carcinogens.  The first is that cancer is disproportionately a disease of the elderly, so that each life "saved" represents relatively few additional years of life.  The death rate for myelogenous leukemia (the type most strongly associated with exposure to benzene), for example, is 26 times higher among people 70 to 74 than among children aged 1 to 5 (Albert et al 1979, table 1).  Thus in evaluating regulations to control carcinogens, we might wish to use a lower value per life saved than in analyzing other programs, such as highway safety, that prevent the deaths of a more representative cross-

section of the population.  We would, in general, prefer to summarize programs in terms of years of life saved.[23] Unfortunately we do not have the data in these three cases to do that.

The second factor is that there is likely to be a substantial delay between when expenditures are made to control carcinogens and when the benefits of reduced risk are reaped. This simply reflects the well-known lags between exposure to carcinogens and the onset of disease.  In benefit-cost analyses, the standard procedure is to discount the streams of benefits and costs to reflect the opportunity cost of the funds employed and time preferences.  Opinion in the economics profession is split as to whether discounting should be applied to health benefits, such as years of life saved.  Most theoretical discussions conclude that discounting is appropriate (e.g., see Raiffa, Schwartz, and Weinstein 1977), but common practice is to ignore the timing issue, thus implicitly not discounting.  (See Page, Harris, and Bruser 1979 for a defense of applying a zero discount rate to risk reductions.)

The impact of discounting is to reduce the relative value of saving lives through control of environmental carcinogens because the substantial lag times between exposure and the onset of cancer mean that the benefits of reducing exposure are reaped many years after the costs are incurred.  We do not have the data to estimate these lags for the case studies, but we believe that they are likely to be large, on the order of a decade or more. If these lags were included, and discounting were applied, the value per life saved would be reduced significantly compared to

programs that have a more immediate impact on fatalities, such as improved fire protection.

**Summary.** The valuation of risk reduction remains uncertain and highly contentious, with little prospect for agreement on any particular dollar value for saving a life. The problem is at least as much one of ethics and politics as it is one of science and the interpretation of empirical evidence. We cannot avoid making tradeoffs between protection and costs, however, whether we do it explicitly or implicitly. Our results are encouraging in that they suggest that precision may not be very important, that many decisions are correct over wide ranges of values. Moreover it appears possible to narrow the range presented in Chapter 2, in particular to reduce the high end. Based on the empirical evidence and the kinds of examples presented above, we find it difficult to justify values much in excess of $1 million per life saved, particularly for airborne carcinogens where there is likely to be a substantial delay before the benefits are reaped and the lives saved are likely to be relatively short.

## IV.  CONCLUSIONS

Our three case studies illustrate many of the problems and uncertainties involved in estimating the benefits of environmental regulation.  They also suggest, however, that while benefit-cost analyses of such regulations never can be very precise, quantitative assessments of benefits can provide invaluable information to regulators interested in improving the efficient use of society's resources.  In this chapter we summarize some of the lessons from the case studies, first with regard to Section 112 of the Clean Air Act and then with respect to the more general use of benefit-cost analysis to evaluate strategies for regulating health-threatening pollutants.

## Section 112

In dealing with "hazardous air pollutants" covered by Section 112 of the Clean Air Act, EPA has followed a technology-based strategy that implicitly treats airborne carcinogens as a homogeneous class, with controls to be set at the BAT level.  The "generic" policy proposed in 1979 would have formalized this approach in an attempt to speed up and routinize the process of listing and regulating such substances. More recently, as discussed in Chapter 1, some members of Congress have proposed forcing EPA to speed up the regulation of Section 112 pollutants, possibly by giving the agency a deadline for making decisions on a list of 37 substances.  Our case studies, however, indicate that airborne carcinogens are a very heterogeneous class, with

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wide variations in benefits (and costs) across substances and source categories. The cases also suggest that the health threat posed by such substances may be relatively modest, so that swift action is not essential to protect public health.

**Heterogeneity.** Even in our small sample of three substances, the benefits of controlling emissions vary enormously because of differences in carcinogenic potencies and in exposure patterns. The estimates in Chapter 2, for example, suggest that each kilogram of coke oven emissions causes, on average, about 500 times as much risk as a kilogram of acrylonitrile or a kilogram of benzene emitted from a maleic anhydride plant. Traditional regulatory analyses that focus on the affordability of controls or costs per unit of emissions controlled would miss these critical differences.

Largely as a result of differences in benefits, the cost per unit of risk reduction also varies greatly across the three cases, differing by more than a factor of 100 between coke plants and the least cost-effective acrylonitrile category. These wide variations suggest that a policy of applying BAT standards to all sources emitting airborne carcinogens will be far from cost effective, imposing higher than necessary costs to achieve any given level of overall risk reduction. Individual substances and source categories need to be considered on their own merits, taking account of potencies and exposure levels as well as technology and affordability.

**Modest Benefits From Control.**  A sense of urgency about the need to control airborne carcinogens is understandable in light of the kinds of facts most readily available.  Recall the information presented in Chapter 1, prior to our detailed examination of the benefits of control.  In both the benzene and acrylonitrile cases, we were confronted with relatively small numbers of sources emitting millions of kilograms of proven human carcinogens each year.  The controls being considered were eminently affordable, with their costs estimated at less than 2 percent of total sales.  Upon detailed examination, however, it became clear that the likely health benefits of the regulations would be very small, less than one cancer avoided per year for the acrylonitrile and maleic anhydride regulations combined.  The result is particularly striking for acrylonitrile, where our estimate is that BAT controls on four source categories would save about one life every five years.  Moreover, for the reasons discussed in Chapter 3, we have reason to believe that these estimates are biased upwards, that the actual benefits probably would be even smaller.  The coke oven standards might provide substantially larger benefits -- perhaps ten fewer cancer deaths per year -- but even there the gain in public health seems rather modest for standards that apply to a major industry on a nation-wide basis.

An important reason for the modest benefits in all three cases is that many sources already have taken action to reduce emissions.  In part these existing controls reflect the firms' own economic interests; benzene and acrylonitrile are valuable substances in their own right and coke oven emissions produce

salable byproducts.  In many cases emissions are subject to control under state regulations or, in the case of coke ovens, OSHA rules designed to protect workers. As a result of these factors, and others, the incremental benefits of regulation under Section 112 are modest.  We cannot be sure, of course, that all Section 112 regulations would yield similarly small benefits. The case studies, however, cast doubt on the proposition that control of airborne carcinogens is likely to lead to major reductions in the nation's cancer burden.  The fact that our cases represent substances assigned relatively high priority by EPA, as indicated by the commitment of substantial agency resources to developing standards, reinforces this skepticism.

## The Role of Benefit-Cost Analysis

As we noted at the outset of this report, many observers believe that massive uncertainties in estimating benefits render benefit-cost analysis an impractical tool for evaluating environmental regulations.  Faced with uncertainties in the risk estimates that span orders of magnitude and the inability to secure agreement as to how to value risk reductions, such critics argue, it is foolish to waste time and resources attempting to perform quantitative analyses. Our results, however, suggest some more positive conclusions:

1.   Despite the great uncertainties, it may be possible in many cases to determine with reasonable assurance whether proposed regulations yield positive net benefits.

2. Detailed analyses of benefits and costs can indicate ways in which regulations can be modified to increase the return on resources devoted to environmental protection.

3. Useful, albeit crude, analyses can be performed relatively cheaply and quickly using data already gathered by EPA.

We elaborate on each of these points below.

**Evaluating Proposed Regulations.** As we discussed in Chapter 3, many of the components in benefit estimation are highly uncertain. Because the final estimate typically is a multiplicative function of these individual components, the overall level of uncertainty is extremely high. Nevertheless, robust conclusions often can be drawn. Most of our findings in Chapter 2, for example, do not depend on whether one accepts the linear dose-response model or a less conservative alternative, or on whether one believes the appropriate value per life saved is $250,000 or $5 million. We make no claim that existing methods of quantitative assessment can yield clear answers in all, or even most, cases. They can, however, help regulators avoid imposing some regulations for which the benefits are far smaller than the costs. Benefit-cost analyses also can identify regulations that clearly provide positive net benefits, though such conclusions may be difficult to draw with great confidence given the number of scientists who believe that dose-response functions exhibit thresholds or are nonlinear.

**Improving Regulations.**   Most discussions of benefit-cost analysis focus on its role as a "test" for proposed regulations. We believe that it is likely to be even more useful as a tool in designing regulations.   In all three cases we were able to find less stringent controls that yielded most of the benefits of the BAT standards at far lower cost.   Although it was not clear that any of these modified uniform standards would yield positive net benefits, it was clear that they were more efficient than the original BAT standards.   Presumably if benefit-cost principles were applied earlier in the regulatory process and used to guide the selection of control options for detailed analysis, larger gains could be reaped.

The case studies also indicated that efficiency could be increased even more by exploiting differences across sources in the marginal benefits of control. These differences arise primarily because of differences in population densities around plants; the public health benefits of controlling emissions are far larger in cities than in lightly populated rural areas. In all three cases, restricting standards to areas where the marginal benefits of control are relatively high led to impressive efficiency gains over uniform standards.

**Information Requirements and Delay.**   An important characteristics for any analytic technique designed to aid in decision making is that it not require data that are unduly expensive or time consuming to obtain.   Analysis is not free; it consumes scarce resources that could be used for other purposes and may cause delays in the regulatory process.   We believe,

however, that a great deal can be done with information that is already collected by EPA. It is important to reemphasize that our analyses of the three case studies are all based on EPA data. Virtually all of the data were drawn from published documents or from contractors' reports. The two exceptions were the cost data for coke ovens, where we obtained an updated computer printout from EPA's contractor, and the exposure estimates for acrylonitrile, which we obtained from another EPA contractor. Thus, performing the kinds of analyses presented here should not significantly increase either the costs or the delays of the regulatory process itself.

We see several areas where additional or improved information might prove very cost effective. The first, as already discussed, would be cost and emission-reduction estimates for a wider range of control options. A sequential strategy probably would be appropriate, with crude and simple analyses of several options followed by more detailed examinations of the most promising ones. We suspect that contractors already collect much of the necessary information, but often the reports do not break down the costs and benefits of individual control components.

The second area where information might be improved at relatively low cost is the exposure estimates. The use of more plant-specific data (especially local meteorological data) should increase the accuracy of benefit estimates at relatively low cost. This would be especially useful if the agency were to adopt a strategy of varying standards in response to differences across plants in benefit levels.

The final area is techniques for estimating unit-risk factors. This is a generic problem of great magnitude, one that is unlikely to be resolved in the forseeable future; we do not have any illusions that scientists will agree on the "best" model to use for low-dose extrapolation or on how animal data should be used to estimate human risk. We think it important, however, to develop techniques for estimating the expected level of risk as well as "high" estimates based on conservative assumption. Such an effort would have to include eliciting from scientific experts the subjective probabilities they assign to the correctness of alternative assumptions and models.

Refining information in the areas outlined above would improve the accuracy of benefit-cost analyses. We stress again, however, that it is likely that decisions often can be made with existing data. Indeed we believe that adoption of benefit-cost principles might reduce the amount of information required in many cases. Current efforts, for example, typically include studies of the "economic impact" of regulations, attempting to predict their effects on plant closings, product prices, and the like. From a benefit-cost perspective, however, such impacts are of second-order importance relative to the direct benefits and costs of control. Application of benefit-cost principles in allocating agency resources also may reduce the costs of analysis by leading to the curtailment of the regulatory process before large expenses have been incurred to gather data. The acrylonitrile case provides an excellent example; we suspect that some crude analysis earlier in the regulatory process -- based on

the unit-risk factor, existing levels of control, and average
exposure factors -- could have indicated the minimal potential
benefits involved, thus eliminating the need for detailed
analyses of control technologies and costs.  Often it is much
easier at early stages to make rough estimates of the benefits
than to predict what the costs of regulation will be.

## Summary

Pleas for the use of benefit-cost analysis in environmental
decision-making are commonplace.  Existing statutes do not
require that such analyses be performed, however, and indeed most
have been interpreted as forbidding EPA from relying on benefit-
cost criteria.  The contribution of this report is to illustrate
in three detailed case studies precisely how benefits assessment
might be employed to evaluate individual regulations, to identify
promising alternatives, and to evaluate the robustness of
regulatory choices to uncertainties.  Although our case studies
relate to a particular statute regulating airborne hazards, we
believe that the conclusions regarding the usefulness of benefit-
cost principles apply more generally.

It is important, however, to put the advantages of benefit-
cost principles in perspective.  A benefit-cost analysis of an
environmental program is not a substitute for good science or
good judgment.  To the contrary, explicit estimation of the
health risks at stake and of the ability of controls to reduce
those risks provides a context for incorporating both science and
judgment into regulatory decisions. Cruder rules based upon
evidence of carcinogenicity or technological feasibility of

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control hide the real choices involved in regulating health-threatening substances, and ultimately are likely to reduce the protection that regulations provide.

**NOTES**

1.  See Nichols (1981 and forthcoming) for an examination of how cost-based and benefit-based reforms may be combined. His analysis, which is primarily theoretical, also includes a case study of benzene and a discussion of how "exposure charges" could be applied to Section 112 pollutants.

2.  Doniger (1978) and Currie (1980) provide useful overviews of the implementation of Section 112.

3.  The benzene case study is based on Nichols (1981). In this paper, however, for consistency with the other case studies, costs have been updated from 1979 to 1982 dollars (using the **implicit** GNP price deflator) and exposure data are reported in **ug/m$^3$-person-years** rather than ppb-person-years.

4.  See Haigh (1982) for more detailed discussions of the coking process and of the health-effects data. The numerical results in this report, however, are not comparable to those in Haigh, as they are based on more recent EPA data.

5.  We have been unable to obtain a detailed breakdown of the status of individual plants. Phillip Cooley of Research Triangle Institute, the primary EPA contractor for the coke oven analyses, however, has told us that the cost data they supplied had positive entries only for plants that are expected to require controls if standards are promulgated.

6.  The acrylonitrile control costs were estimated using data from several sources. See chapter 2, in particular the notes for table 2.11.

7.  Sales of maleic anhydride were $142 million in 1979 (calculated from U.S. EPA 1980 and Chemical and Engineering News, June 13, 1983). Sales of coke were $4.9 billion in 1980 (calculated from Bingham et al. 1982). Sales of the four acrylonitrile categories were $1.3 billion in 1979 (Energy and Environmental Analysis 1981); estimated control costs in 1979 dollars are less than 2 percent of that sales level.

8.  E.O. Order 12291 lists several criteria for defining a "major" rule: (1) an annual effect on the economy of more than $100 million; (2) "major" increases in prices; or (3) "significant" adverse effects on competition, employment, investment, productivity, or innovation (Environmental Law Reporter 1981, 10044).

9.   Schelling (1968) generally is credited with being the first
     to argue that willingness to pay for risk reduction is the
     appropriate conceptual approach to valuing "life saving."

10.  The technical terms for these two measures are "compensating
     variation" (CV) and "equivalent variation" (EV). In
     general, when discussing risk reductions, EV (how much an
     individual would have to receive to be willing to accept in
     lieu of the risk reduction) will exceed CV because of income
     effects.  For small changes in risk, however, the
     differences between the two measures will be negligible.

11.  The primary source of data for this case study is U.S. EPA
     (1980).  For additional sources, see Nichols (1981).

12.  The primary sources of data for the coke-oven case study are
     U.S. EPA (1981a, 1981b, 1982, and 1983b). Updated
     compliance costs provided by Research Triangle Institute
     (1983).

13.  This minimum-cost allocation could be achieved by levying a
     uniform charge on exposure (not emissions) or a system of
     plant-specific emission charges that vaaried in proportion
     to exposure factors.  See Nichols (forthcoming) for a
     discussion of exposure charges and applications to benzene
     emissions from maleic anhydride plants.

14.  The data for the acrylonitrile case study were assembled
     from several sources, including Click and Moore (1979), Key
     and Hobbs (1980), Energy and Environmental Analysis (1981),
     Radian Corporation (1982), Albert et al. (1982), and Suta
     (1979, 1982a, and 1982b).

15.  The estimated **reduction** in emissions from controlling those
     plants is 312,000 $ug/m^3$**-person-years,** while the estimated
     cost is $8.4 million.

16.  The estimated **reduction** in exposure from controlling that
     plant is 98,000 $ug/m^3$**-person-years,** while the estimated cost
     is $0.8 million.

17.  The estimated reduction in emissions from controlling door
     leaks at that plant is 20,700 kg.  With an estimated
     exposure factor of 6.1, **that** translates to a reduction. in
     exposure of 126,000 $ug/m^3$**-person-years.**  Applying the CAG
     risk factor of 1.3 x $10^{-5}$ implies that 1.6 lives would be
     saved, as compared to 0.6 for the maleic anhydride and
     acrylonitrile BAT standards combined.  The estimated control
     cost for that one coke plant is $184,500, as compared to
     over $31 million for the maleic anhydride and acrylonitrile
     standards.

18.  The estimated cost for those four plants is $1.0 million per
     **year,** while the estimated reduction in exposure is 363,000
     $ug/m^3$**-person-years.**

19. In its preliminary report on benzene, for example, the CAG said that the linear model" is expected to give an upper limit to the estimated risk" (Albert et al. 1977, 1).

20. Nichols (1981, ch. 9) provides equations for the various models and an example of their widely different predictions at low doses when estimated from the same high-dose data.

21. Suppose the reduction in emissions, X, is **distributed** log-normally, where ln(X) has mean m and variance $s^2$. The upper and lower 95 percent confidence limits then will **be** exp(m+2s) and exp(m-2s), respectively. In chapter 2, we used a simple average of these two extremes, 0.5[exp(m+2s) + exp(m+2s)]. If emissions **are** distributed log normally, however, the mean is $\mathbf{exp(m+s^2/2)}$. The ratio **of** the mean to the average of the two limits is then $\mathbf{2exp(s^2/2)/[exp(2s) + exp(-2s)]}$. To calculate this ratio, we need to estimate s **(or $s^2$)**. This can be done using the ratio of the upper and lower confidence limits, R: R = exp(m+2s)/exp(m-2s) = exp (4s). Thus, s = ln(R)/4. For example, if R = 11.2 (the ratio of the maximum to minimum for doors), s = ln(11.2)/4 = 0.604. The ratio of the mean to the average of the limits is then $\mathbf{2exp(0.604^2/2)/(exp[2(0.604)] + exp[-2(0.604)]) =}$ 0.668.

22. For example, EPA estimates that the annual risk of leukemia for the maximum exposed individual residing near an uncontrolled model maleic anhydride plant **is** ₀0.038 per 10,000 (U.S. EPA 1980, app. E), or 3.8 x $\mathbf{10^{-6}}$. This risk is less than 1 percent of the average annual risk of dying in a motor vehicle accident (Wilson and Crouch 1982, 176).

23. Zeckhauser and Shepard (1976) argue that mortality benefits should be summarized in terms of the discounted number of "Quality Adjusted Life Years" (QALYs) save. Their QALY measure adjusts for reductions in the quality of life due, for example, to disability.

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**PART 7**

**BENEFIT-BASED FLEXIBILITY IN ENVIRONMENTAL REGULATION**

David Harrison, Jr.
Albert L. Nichols

## I.   INTRODUCTION

Environmental regulation in the United States relies heavily on emission standards that are uniform for all emitters in broad industry classes.  These uniform standards were the result of a series of laws passed in the early 1970's, most notably the Clean Air Act and the Clean Water Act, which required the Environmental Protection Agency (EPA) to set uniform standards for an industry based upon technical feasibility and the ability of the industry to afford controls.[1]   Such rigidity was needed, it was argued, to show the nation's commitment to environmental goals at a national level, to prevent the EPA from using its discretion to water down the commitment, and to streamline the process of setting environmental controls.

The new wave of environmental laws has had its share of critics, most of whom have focused on its cost.  Many have pointed to the sheer size of the program.  The air and water standards alone are projected to cost more than $700 billion over the decade from 1979 through 1988.[2] Economists and other efficiency-minded critics have emphasized the excessive cost of ignoring differences across sources in the costs of controlling emissions, arguing that overall costs could be reduced with no

sacrifice in environmental quality if regulations were more flexible, imposing more stringent controls on low-cost sources and more lenient limits on high-cost sources. The favorite prescription of economists is emission charges or other incentive-based mechanisms (such as marketable permits) that automatically allocate control efforts in accordance with marginal costs.[3]

The long-standing arguments for cost-based flexibility are beginning to have some impact on policy. Although no full-fledged economic incentive schemes have been implemented, in the past few years the EPA has begun to introduce some cost-based flexibility in its regulations, primarily through limited versions of the marketable permits approach, such as the "bubble" policy, emission offsets in non-attainment areas, and "banking" (del Calvo, 1981). In at least one case (chloroflourocarbons), EPA is considering a relatively "pure" system of marketable permits (Rabin, 1981). These modifications may yield significant gains in efficiency.

Discussions of regulatory reform, however, at both the theoretical and practical levels, generally have ignored another potential source of major gains in efficiency: benefit-based flexibility. Sources differ not only in the costs of controlling emissions, but also in the damages their emissions cause. The link between emissions and damages often varies widely across both time and space. The health risk caused by the emission of a toxic substance from a chemical plant, for example, varies dramatically depending on whether the plant is located in a densely populated city or in a lightly populated rural area. The

damages caused by emissions from a given site may also vary across time, depending on meteorological conditions and other factors.  A regulatory strategy that exploits this diversity -- by requiring stringent controls where benefits are high and relaxing controls where benefits are low -- could yield significant gains over the uniform regulations that now dominate environmental regulation.

This paper evaluates the case for incorporating benefit-based flexibility into environmental regulation. Although the concept is quite general, for ease of exposition most of our analysis focuses on health-threatening pollutants and on varying standards geographically.  We first lay out the theoretical rationale for benefit-based flexibility and then review the empirical evidence on the magnitudes of potential gains. To counter the potential criticism that the concept is fine in theory but unworkable in practice, in section III we lay out a simple strategy for developing benefit-based standards that could be incorporated easily into the EPA's standard-setting process. This system makes several simplifying assumptions, and in Section IV we discuss the major theoretical issues that might compromise the efficiency of our simple system.  Next, we consider some of the distributional concerns surrounding a move to benefit-based flexibility.  In Section VI we discuss how benefit-based and cost-based flexibility can be combined.  The final section presents our conclusions.

## II. THE THEORETICAL CASE

Benefit-based flexibility promotes efficient environment protection in two ways. First, varying control requirements concentrates emission control efforts where marginal benefits are greatest. Second, differential controls provide incentives for firms to select low-damage sites for polluting activities. Both effects lower the cost of reducing damages, although their relative importance will depend upon the nature of the industry being regulated. This section illustrates these two rationales for benefit-based flexibility with a simple theoretical model. The model both makes the case for such flexibility more rigorous and provides a convenient reference point when we consider some of the potential complications of our specific proposal.

### Differential Control

The role for benefit-based differential emission controls is shown easily with the aide of the following model.[4] Suppose that there are n sources, each emitting the same hazardous pollutant. The cost of reducing emissions at the ith plant is $C_i(r_i)$, where $r_i$ is the reduction achieved. We make the usual assumptions that marginal control costs are positive $(C_i'>0)$ and increasing $(C_i''>0)$ at each source. Note that the costs of control may vary across sources. The benefits of control also may vary. In particular, let $E_i$ be the amount of exposure caused by a unit of emissions

from the ith plant, where $E_i$ is a function of the population density and meteorological conditions around the plant. Reducing emissions by $r_i$ at the ith source thus reduces exposure by $r_i E_i$.

For simplicity, we assume that the benefit from control is proportional to the reduction in total exposure. (This is consistent with a linear dose-response model, which is widely assumed for carcinogens. In Section IV we consider how a nonlinear damage function affects our results.) The net benefit of reducing emissions from all sources is given by:

$$N = V \sum_{i=1}^{n} r_i E_i - \sum_{i=1}^{n} C_i(r_i) , \tag{1}$$

where V is the shadow price on exposure. (More specifically, V is the risk per unit of exposure times the dollar value placed on reducing risk.) Differentiating with respect to each $r_i$ and setting the results equal to zero yields the first-order optimality conditions:

$$C_i'(r_i) = V E_i \quad \text{for } i=1,\ldots,n. \tag{2}$$

Equation (2) states the familiar result that the marginal cost of reducing emissions at each source should be equal to the marginal benefit. But this formulation makes the limitations of cost-based flexibility clear; if the exposure factors $(E_i)$ vary across sources, it will not be optimal to equate marginal emission control costs $(C_i'(r_i))$, as would happen with a uniform emission charge or a marketable emission permit system.

Consider a simple example. Two plants, A and B, are identical in every respect (including emission control costs), except that plant A is located in New York City, while B is located in a remote rural area.  As a consequence, A's exposure factor is 100 times that of B.  A uniform emission standard is imposed on both plants.  Because both plants face the same costs, by assumption, the principles of cost-based flexibility are not violated.  No reallocation of control efforts could reduce costs without also increasing emissions.  The outcome, however, clearly is not cost-effective in terms of the appropriate measure of benefits, reduced damages, for while the marginal costs of controlling emissions are the same, the marginal cost of controlling damages is 100 times higher at B than A.  That is, shifting $1 in control expenditures from B to A would have no effect on emissions, but would reduce damages.

Figure 1 illustrates the example.  The marginal benefits of controlling emissions at the two plants are shown, respectively, by the curves labeled $MB_A$ and $MB_B$.  The marginal cost of control is shown by the curve MC.  The optimal uniform standard is $r_0^*$, the point at which MC intersects $MB_0$, the average of the two individual plants' marginal benefit curves.  With benefit-based flexibility, however, net benefits can be increased by tightening the standard at A to $r_A^*$, and loosening it to $r_B^*$ at B.  The increase in net benefits is equal to the sum of the two shaded areas.
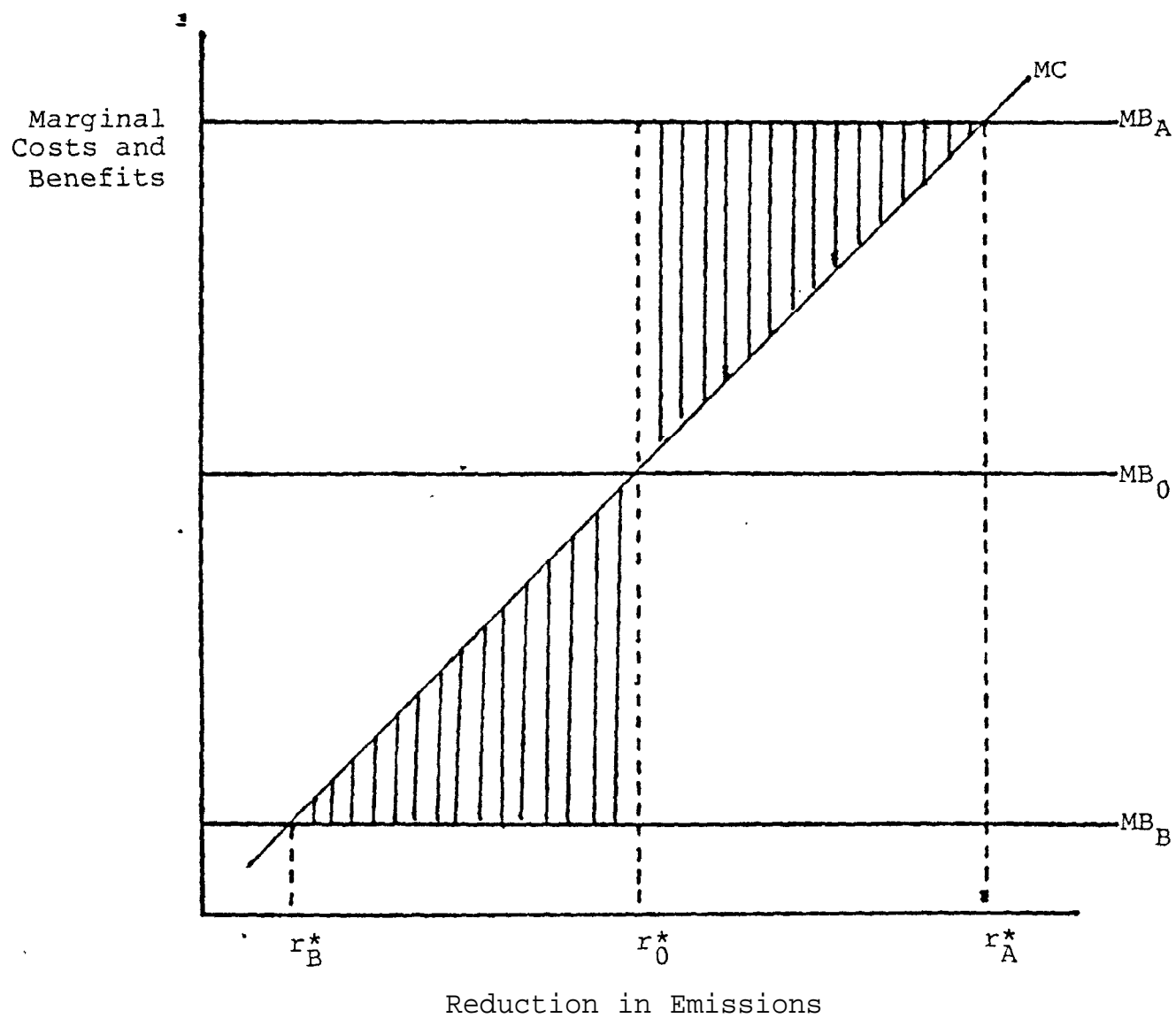
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Figure 1.   Gains from Benefit-Based Flexibility

The gain in efficiency and the optimal degree of differentiation under benefit-based flexibility depend on a variety of factors. Such flexibility obviously is more important if the differences in marginal benefits are large. It is less important if marginal costs vary sharply as emission levels change; in that case the optimal degree of control will not differ significantly across plants, as Figure 2 illustrates. The marginal benefit curves are identical to those in Figure 1, but the marginal cost curve is much steeper. As a result, the optimal benefit-based standards at the two plants are almost the same, and the net gains due to benefit-based flexibility are smaller than in the previous figure.[5]

This example ignores one potentially serious complication. We have assumed that the two sources have the same control cost schedules. As a result, tighter controls are imposed on the source with the higher exposure factor. In some cases, however, exposure factors and costs may be positively correlated, with high-exposure sources also having high control costs. If this effect is strong enough, it may be optimal to impose less stringent controls on the high-exposure (high-cost) sources. Although we suspect that such cases are rare, we discuss ways of dealing with them in Section IV.

## Location Incentives

Our simple model assumes that source locations are fixed, so that the gains under benefit-based flexibility are due solely to differential control levels. When that assumption is relaxed, siting becomes a potential tool for reducing damages. In our
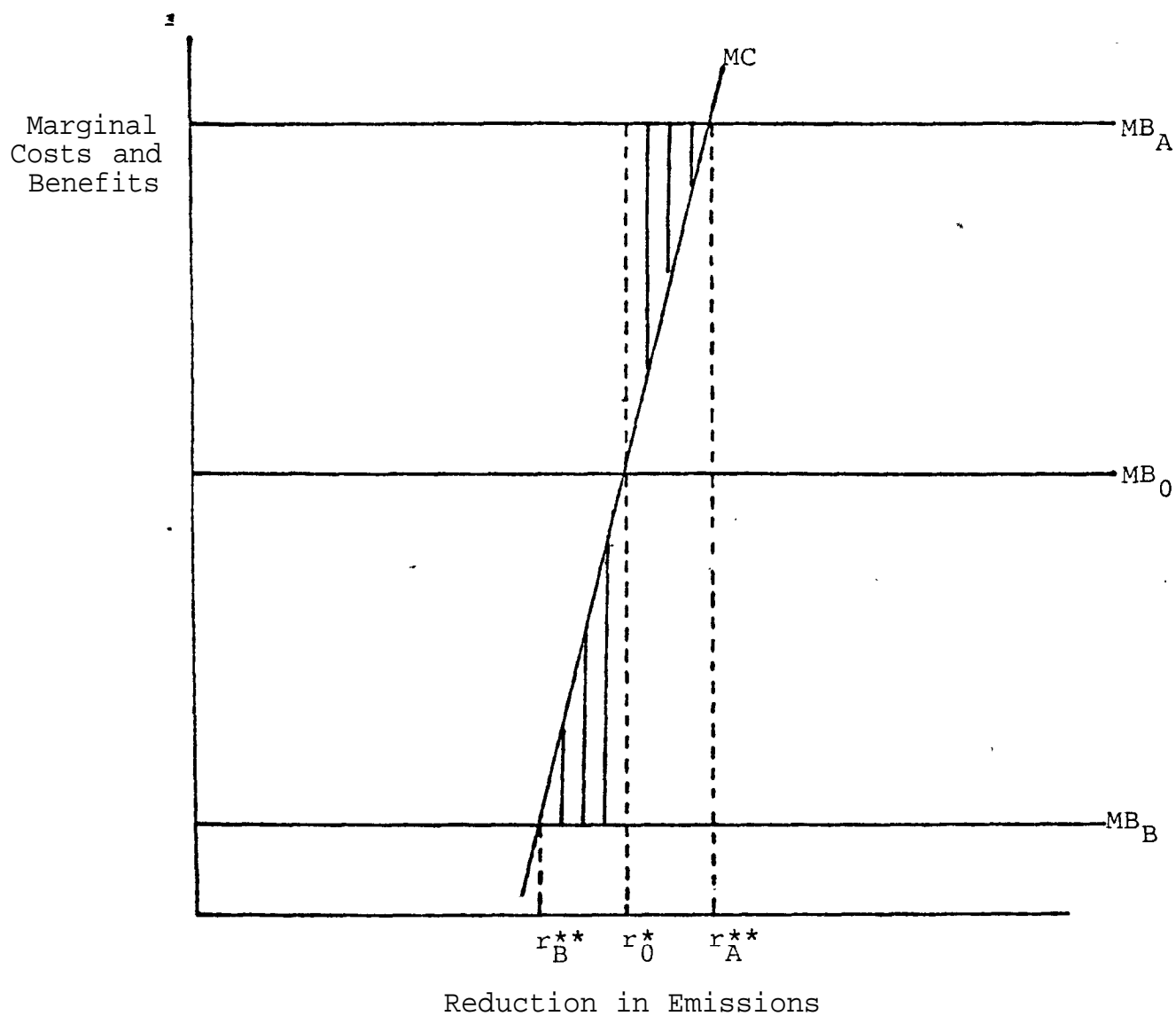
Marginal
Costs and
Benefits

MC

MB$_A$

MB$_0$

MB$_B$

$r_B^{**}$    $r_0^*$    $r_A^{**}$

Reduction in Emissions

Figure 2.   Gains from Benefit-Based Flexibility
with Sharply Rising Costs

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hypothetical case, for example, moving plant A to plant B's site would generate the same benefits as 99 percent control of A's emissions  Benefit-based flexibility, by imposing more stringent requirements on sources at high-damage sites, encourages firms to locate noxious facilities farther from heavily populated areas. In contrast, under uniform emission standards (or emission charges), firms have no incentive to consider damages in their siting decisions.

The incentive that differential standards provide for low-damage siting is illustrated in Figure 3. As before, MC is the marginal cost of controlling emissions and $MB_A$ and $MB_B$ are the marginal benefits of emission control at two sites.  The maximum level of emission control (zero emissions) is $r_M$.  Under a benefit-based strategy, standards would be set at $r_A$ for a plant at site A and at $r_B$ for a plant at site B.  Moving the plant from site A to site B would reduce control costs by the area a+c. Thus, a firm would move from A to B if the costs of the move (taking into account all of the advantages of site A relative to site B as well as the moving costs themselves) were less than a+c.

Although differential standards provide more efficient location incentives than a uniform system, they do not necessarily provide precisely the right incentives.  The full welfare changes associated with a move from A to B include the difference in damages, equal to b-c, as well as the reduction in control cost (a+c).  Thus, moving to site B would increase net benefits if the costs of the move were less than a+b. As the
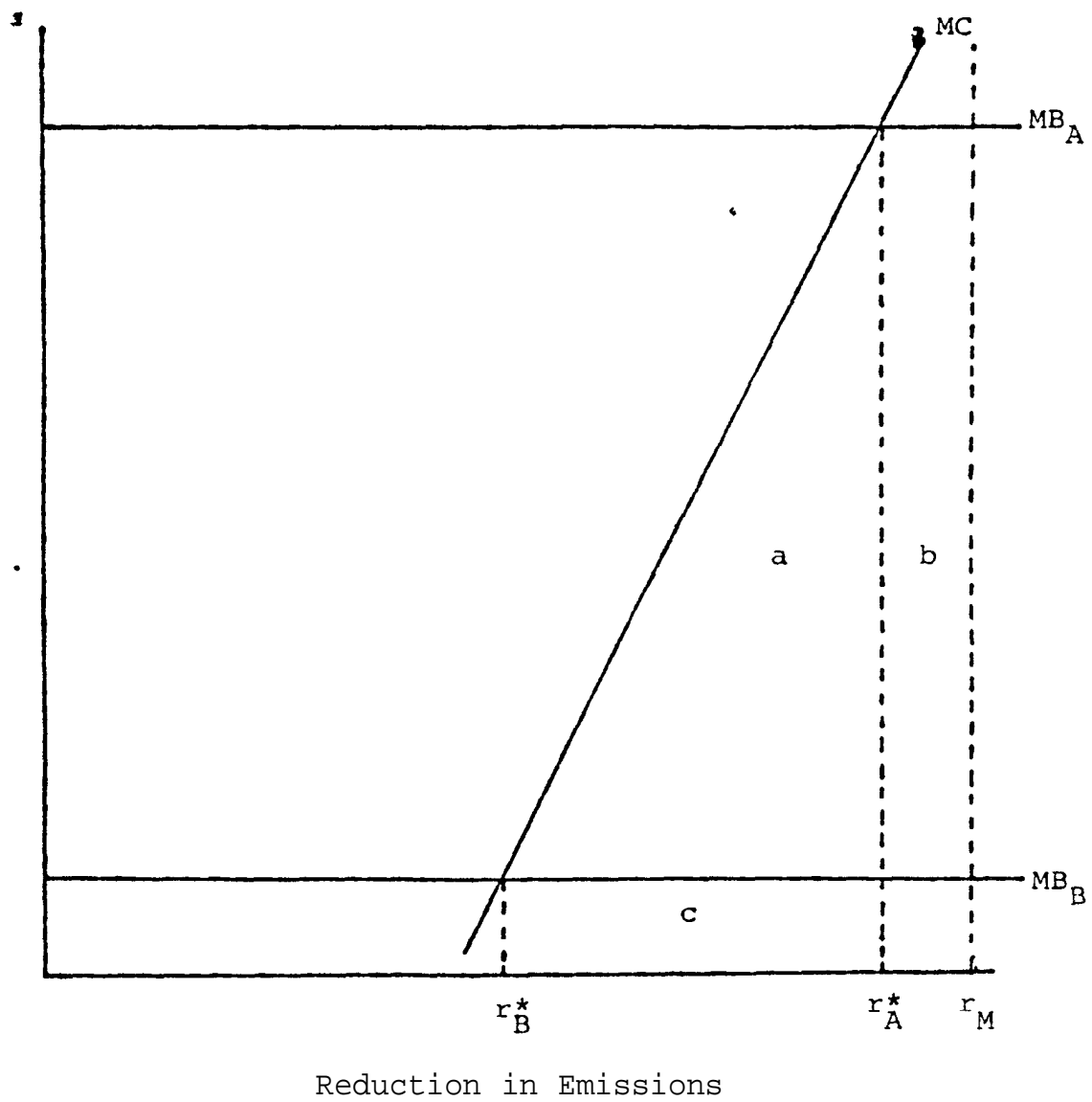
Figure 3.  Siting Incentives with Benefit-Based Flexibility

firm compares moving costs to a+c under the differential standard, the incentive to move will be inadequate if c is less than b and excessive if c is greater than b.[6]

The importance of the location incentives under benefit-based flexibility depends on the feasibility and costs of low-damage siting, which are likely to vary widely by source. Differential standards should have little effect on the locational pattern of sources owned by individuals for personal use; tighter standards for automobiles in urban areas, for example, almost certainly would not prompt many people to move to rural areas. (Price increases due to the stricter standards might result in somewhat lower levels of auto ownership in such areas, however.) Similarly, firms that sell directly to consumers have limited flexibility in choosing their locations; few service stations would move from Los Angeles, for example, if stringent vapor recovery regulations were imposed in that metropolitan area.

Locations are more flexible, however, for many industrial sources, particularly new ones. Companies often consider several sites when deciding where to build a new plant. Many factors enter into these decision -- including wage scales and other aspects of local labor markets, transportation costs to final markets, access to raw materials, and state and local taxes -- so differences in environmental regulations are unlikely to be dominant determinants in most cases. They could, however, tip the balance in favor of low-damage sites when the other factors are roughly balanced. Differential regulations are likely to be more important for siting decisions in pollution-intensive

industries, where emission control costs could be a significant fraction of total costs. For such sources, low-damage siting may offer a cost-effective strategy for reducing damages.

## III.  EMPIRICAL EVIDENCE

The theoretical case for incorporating benefit-based flexibility in environmental regulations is straightforward. Theory does not, of course, indicate the size of the efficiency gains -- whether benefit-based flexibility is an important element of reform or a minor refinement of little practical interest. Compared to the interest in cost-based reforms, relatively few studies have estimated the costs and benefits of such strategies, but they suggest that adjusting the stringency of standards to take benefits into account can generate large efficiency improvements, exceeding in some cases the gains from cost-based flexibility.

The relevant studies may be grouped into two categories.[7] The first and most important consists of studies that estimate the advantages of allowing federal standards to vary geographically.  The second consists of evaluations of the importance of siting in reducing the environmental impacts of major projects.

### Federal Programs

Table 1 summarizes six studies that estimate the effects of geographic variation in national environmental programs. The studies range across media, including air pollution, water pollution, and aircraft noise. The authors of each study have

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Table 1.  Empirical Studies of Benefit-Based Flexibility

| Author (year) | Pollutant or source | Number of zones | Percentage Decrease from uniform base | |
|---|---|---|---|---|
| | | | Costs | Benefits |
| Harrison (1975) | auto air pollution | 2 | 35 | 9 |
| Luken et al. (1976) | water pollution | 2 | 73 | 0[a] |
| Harrison (1983) | aircraft noise | 3 | 50 | 18 |
| Nichols (1983) | airborne benzene | 2 | 63 | 4 |
| Haigh (1982) | coke oven emissions | 2 | 54 | 19 |
| Perl and Dunbar (1982) | coal-fired power plants | 21 | 62 | 14 |

Notes:

[a]assumes no benefits from further reducing discharges into basins that are already "clean."

divided the country into two or more classes based on differences in the benefits of pollution control, and then examined alternative standards for each class.

None of the empirical studies provides the ideal comparison, which would measure the gain in net benefits from switching from an optimal uniform standard to an optimal benefit-based variable standard (as illustrated in Figure 1). All studies use the current uniform controls as a baseline, and then estimate the cost savings and benefit reductions that result when standards are relaxed in low-benefit areas. There is, therefore, no guarantee that the comparison is between optimal representatives of either the uniform or the benefit-based flexible strategies. These studies also fail to estimate the gains that might arise from relocating sources to lower-damage sites. Nevertheless, they provide an indication of the potential gains from benefit-based flexibility.

Harrison (1975) compares federal new car emission standards, which are the same for all cars regardless of where they are driven with a "two-car" strategy that would loosen the standards outside of the most heavily polluted and densely populated urban areas. The benefits of controlling automotive emissions are small in rural areas and smaller cities both because the air is already relatively clean and because lower densities mean that fewer persons are affected by a given car's emissions. Harrison estimates that switching to a two-car strategy would reduce long-run costs by 35 percent, with only a 9 percent reduction in benefits, as measured by the average reduction in exposure to three pollutants.

Luken et al. (1976), in a study done for the National
Commission on Water Quality, evaluate several alternatives to
stringent controls on industrial and municipal water pollution
sources.  The 1972 Amendments to the Water Pollution Control Act
mandated two stages of clean-up requirements, a first stage to be
accomplished by July 1, 1977, and a more stringent stage to be
reached by July 1, 1983.  The results in Table 1 are based on
Luken et al.'s calculations of the effects of maintaining
national Stage 1 standards, but eliminating the Stage 2 standards
for water basins with "good" water quality after Stage 1.  The
Stage 2 control costs are reduced by 73 percent under the
benefit-based approach because 78 of the 99 river basins achieve
good quality after Stage 1.  The authors did not calculate
benefit measures for the alternative.  It is plausible, however,
that the stringent Stage 2 controls generate virtually no
additional benefits in the "good quality" basins, as most water
pollution control benefits are accounted for by increased
recreational use, which would not be affected by additional
clean-up where water quality is already good.

Current standards require all aircraft to meet the same
noise standards regardless of where they are flown.  But it is
clear that a given takeoff or landing causes much more annoyance
at airports like Boston's Logan Airport, which is located in a
densely populated area, than at an airport like Dallas-Ft. Worth,
which is located in a low density rural-suburban area.  Harrison
(1983) estimates the costs and benefits of permitting more
lenient standards for airports classified as "moderate benefit"

or "low benefit."  After making allowance for the impossibility
of precisely matching aircraft types to even these broad airport
categories (and thus the need for a greater number of stringently
controlled aircraft), Harrison estimates that such a scheme
could reduce overall compliance costs by 50 percent while
reducing benefits (as measured by the number of people no longer
exposed to high noise levels) by only 18 percent.

Two studies have examined the use of benefit-based
flexibility in regulating toxic air pollutants. Section 112 of
the Clean Air Act gives the EPA authority to regulate both new
and  existing sources of "hazardous" air pollutants.  Nichols
(1983) evaluates options for regulating, airborne benzene,
focusing on the importance of variability in marginal damages
caused by emissions.  For one category of benzene emission
sources, maleic anhydride plants, the estimated exposure factor
(population exposure to benzene per unit of benzene emitted)
varied by a factor of almost 50, although only 8 plants were
involved.  Nichols estimates that if the four plants with lower
exposure factors were exempted from EPA's proposed uniform
emission standard, costs could be cut 63 percent while reducing
benefits only 4 percent.  Although he does not calculate the
costs and benefits of regulatory alternatives for other sources
of benzene emissions, Nichols reports that exposure factors vary
by more than 150 for both automobiles and service stations;

Using a framework similar to that of Nichols, Haigh's (1982)
study of coke oven emissions provides further evidence of the
importance of benefit-based flexibility.  He estimates that
exposure factors vary by more than a factor of 150 across the 58

plants studied by EPA. Although EPA has not formally proposed a coke-oven regulation, Haigh identifies a uniform control requirement that he considers a plausible estimate of what EPA might require if it decides to regulate under Section 112. An alternative that exempts all but the ten highest-exposure plants from the regulations would cut costs by 54 percent, while reducing benefits by only 19 percent.

Perl and Dunbar (1982) evaluate alternatives for controlling sulfur dioxide emissions from coal-fired power plants. They estimate that the marginal benefits of lower sulfur dioxide emissions vary across 21 electricity demand regions by more than a factor of three; the marginal health benefits alone vary by about a factor of 150. An optimal system of region-specific $SO_2$ taxes, they estimate, would cost 62 percent less than the current regulations, while reducing benefits only 14 percent. Unfortunately, Perl and Dunbar do not report sufficient information to estimate how much of the cost saving is due to cost-based flexibility and how much to benefit-based flexibility.

## Remote Siting

Numerous environmentally harmful facilities can be placed in different locations -- airports; power plants, and highways are common examples -- to reduce damages. Ideally, one would want estimates of the reduced damages with remote siting, the savings in control costs if different controls were required, and any increase in non-environmental costs due to remote siting. No studies provide such a full set of estimates of the savings

possible from remote siting as an alternative to stringent controls, but several discuss siting as a tool for reducing risk.

Yellin and Joskow (1979) examine the merits of remote siting of nuclear-powered generating plants. They argue that the Nuclear Regulatory Commission has paid insufficient attention to the potential consequences of major releases of radiation from reactors, instead focusing almost exclusively on standards to reduce the probability of such accidents. Yellin and Joskow show how remote siting can reduce those consequences, albeit at the cost of longer transmission lines and greater transmission losses. Their argument is not for differential regulations based on siting, but rather for adding siting as another method of control. Unfortunately, they do not provide quantitative estimates of the greater transmission costs or the lower control costs that remotely-sited nuclear power plants might incur.

Wilson et al. (1980) argue that siting policy also can play a significant role in reducing the damages caused by coal-fired power plants. They suggest that such plants be located, when possible, in lightly populated areas downwind of major population centers. The Northeastern seacoast offers particularly attractive sites from this perspective, as prevailing winds would blow most of the emissions out to sea. The authors estimate that such siting could reduce damages by a factor of 3 to 32, depending on the time of year and the rate at which the sea absorbs sulfates. Again, they provide no estimates of increased transmission control or decreased emission control costs if such a policy were adopted.

Lathrop and Linnerooth (1982) discuss another example of using low-damage siting as an adjunct to more conventional controls of hazardous activities:  liquified natural gas (LNG) transfer facilities.  California requires stringent safeguards on such facilities, but also restricts them to sites with a population density of less than ten people per square mile. Unfortunately, the authors provide no quantitative estimates of either the benefits or the costs of the two means of reducing the risks from LNG.

## Conclusions from the Empirical Studies

These studies suggest two major conclusions.  First, the gains from introducing benefit-based flexibility are likely to be significant, quite possibly on a par with the more widely studied advantages of cost-based flexibility.  Most of the increase in efficiency probably would come from differential controls, in particular relaxing controls from current levels where the marginal benefits are small.  Although the empirical evidence is much sketchier, it appears that damage-sensitive siting also may be an important tool for risk reduction for some types of hazardous facilities.

Second, large gains appear possible even with relatively crude benefit-based strategies.  Four of the six studies of Federal regulations differentiated only two benefit classes, and a fifth used three classes.  Moreover, in most cases the studies were able to consider only a very restricted range of control

options, typically an existing (or proposed) standard and one or two alternatives.  Enlarging the numbers of classes and control options presumably would permit larger gains.